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The role of riparian buffer management in reducing off-site impacts from grazed dairy systems

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Abstract

Agriculture in general and intensive animal production in particular can degrade the environment, especially as a consequence of the overuse of nutrients. Intensively grazed dairy systems, defined by the presence of foraging animals in the landscape, are often considered a more benign approach to dairy production with perceived smaller impacts due to the reduced requirement for manure disposal. However, grazing dairy cows contribute nutrients and pathogens in excreta, and sediment through landscape deformation. These impacts can dramatically increase in parts of the farm such as feeding, watering and sacrifice areas, laneways and night paddocks where animals are concentrated and spend a disproportionate amount of time. Other practices such as the disposal of dairy shed or dairy factory effluent and cultivation of fodder crops can also pollute the environment. A common approach to reduce nutrient, pathogen and sediment losses from dairy farms is to establish buffer zones in riparian areas that act as an interface between upland land use and waterways. This is generally done by fencing riparian areas to exclude stock and revegetating with understory and overstory species, with the aim of increasing infiltration, trapping sediment and decreasing contaminant losses from upland pastures. However, poorly designed and managed riparian areas may themselves contribute to further environmental degradation. Rarely is an integrated approach, including factors such as animal behavior and dairy farm management practices, as well as an analysis of landscape and riparian hydrology used in developing riparian management recommendations for individual farms. This paper reviews the threats posed by intensively grazed dairy systems, approaches to improve riparian zone management and recommends future research needs.

Key words: nutrients, phosphorus, nitrogen, fecal organisms, sediment, filter strips, watershed

Introduction

Milk production per cow, farm size and herd numbers continue to increase in most of the dairy-producing regions of the industrialized world despite declines in farm numbers[1]. This intensification of the dairy industry has been in large part due to greater use of inputs such as fertilizer and feed supplements which have contributed in many instances to nutrient surpluses on farms[2–5]. These on-farm nutrient surpluses can have negative consequences for the environment, including emissions through various nutrient transport pathways[6,7]. The factors attributed to intensification of the dairy industry and the consequences of the intensification are in general similar for grazing and confinement systems[8]. Manure management of housed animals is a priority in most confinement dairy systems. However, despite the perception that grazed dairy systems are more environmentally benign than confinement systems[9,10], animals defecating and urinating across the landscape will result in different pollutant sources and propensity to degrade waterways.

As a mitigating measure farmers are actively encouraged to improve riparian management, firstly by fencing these areas to exclude stock, and also by revegetating fenced areas to reduce contaminant delivery to streams and improve native biodiversity attributes. However, livestock movement and management, both within the riparian area and across the broader landscape, challenge traditional approaches to riparian management.

This review describes the impacts posed by grazing dairy farms on contaminant losses, discusses the role of riparian buffers within these systems in temperate regions

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and presents options for improved management to reduce their potential environmental impact. The review draws on data primarily from Australia and New Zealand, where the majority of dairy farms are grazed pasture systems, as well as from elsewhere where relevant.

Characteristics of Grazed Dairy Systems

Grazed dairy systems (such as in Australia, New Zealand or Argentina) depend on high-quality permanent pastures to support the majority of the nutritional needs of lactating dairy cows; in contrast to the predominantly confinement systems in Europe and the USA[11,12]. In many European dairy systems grazing has traditionally been in summer[13,14], whereas in the USA grazing is largely associated with extensive livestock systems[15].

Typically, grazing dairy farms are divided into 'paddocks' that are fenced with at least one gateway at one end to allow entry and egress, and which often contain a usually immobile watering point or trough (Fig. 1). Pastures are usually dominated by grasses and may also contain varying proportions of a legume. Pasture productivity can be limited by temperature and water, both of which contribute to the temporal pattern of pasture production for a particular region[13,16].

Greater nutrient supply can significantly boost pasture growth and productivity. In Australia and New Zealand the broad-scale use of phosphorus (P) and potassium (K) fertilizers has been a common practice to overcome naturally occurring nutrient deficiencies[17,18]. While nitrogen (N) was traditionally supplied to pastures through biological fixation by the legume–*Rhizobium* symbiosis, the application of nitrogenous fertilizers is now widely used to increase pasture production. Similarly, in Europe, where P, K and sulfur (S) deficiencies are less common, N fertilizer has been extensively used since the middle of the 20th century[13].

Grazing pastures on a rotation, the length of which is determined by pasture dry matter on offer, increases pasture utilization and thus milk production and profitability[16]. To maximize pasture utilization farmers often 'strip-graze' paddocks, where cows are provided with access to narrow strips of pasture within individual paddocks. The strips are usually not back-fenced, allowing cows to revisit previously grazed strips[10], while ensuring cows have access to immobile water troughs.

Despite the cost advantage of pasture as the major feed source, the grazing industries have increased their use of supplemental feed substantially over the past two decades to meet feed gaps and further increase milk production[19,20]. Feed supplements (such as grain, grain-based pellets with minerals, purchased forages such as hay and silage and increasingly by-products such as palm kernels, citrus pulp and molasses) are used to manage temporal variations in pasture availability, and to balance energy, protein and mineral intakes[13,20]. Thus, more than 90% of

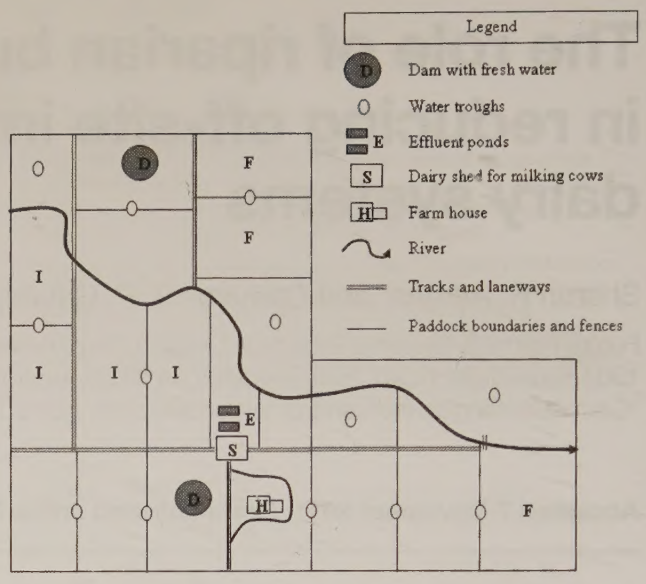

Figure 1. Schematic representation of a grazed dairy farm showing typical management activities. The activities include the dairy shed (S) where the cows are milked at least twice each day; effluent ponds (E) for storage of dairy effluent (cow excreta deposited in the dairy shed during milking combined with the wash down water); farm dams (D) where water is stored for use on the farm. Pastures are subdivided into paddocks that are part of the grazing rotation and are visited by cows throughout the year. Some of the paddocks (F) may be taken out of the grazing rotation during the spring months for forage conservation (hay or silage), for renovation of pastures, or for growing summer fodder crops (e.g., turnips, maize). Some paddocks (I) will be irrigated with effluent from the storage ponds, usually during the dry summer months.

Australian dairy farms regularly purchase cereal grain-based supplements of greater than 1.1 tonne $cow^{-1} yr^{-1}$, and forage inputs ranging from 0 to 1.4 tonne $cow^{-1} yr^{-1}$ on the average dairy farm[21]. The increased reliance on purchased feed, in addition to increasing fertilizer rates, has generally resulted in greater nutrient inputs and net positive balances for N, P, K and S on grazed dairy farms, leading to soil fertility levels on many paddocks well in excess of that required for pasture growth[22].

Grazing cows move frequently by means of an interconnecting network of laneways and tracks between paddocks and to the dairy shed where they are usually milked twice daily. In the past, as well as a source of water, waterways were also used for the disposal of dairy shed effluent, and consequently dairy sheds are often located adjacent to or within the riparian zone. To prevent the direct contamination of surface and ground water, farmers are encouraged to collect deposited feces and urine as well as waste milk from concreted areas where cows are held, fed and milked. This effluent, usually stored in earthen-constructed ponds[23], is often used to irrigate pastures (most commonly those close to the dairy shed) during the dry summer period to take advantage of

the nutritive value[24]. Application of dairy shed effluent generally increases soil P, K and N, although inconsistencies were observed in the different experiments reviewed by Hawke and Summers[25].

Management of grazed dairy farms often includes the use of 'night paddocks' in which the cows are held between the evening and morning milking, and which are generally located near to the dairy shed. Farms may also have other high animal density areas such as feed pads, yards where cows are held prior to or just after milking, or small paddocks where cows are kept during calving or when in need of special care. Farmers also conserve excess pasture in selected paddocks either as silage (spring) or hay (late spring, early summer). Summer fodder crops such as turnips (*Brassica* spp.), maize (*Zea mays*), and millet (*Echinocholoa utilis*) may also be grown to provide additional forage when pasture growth may be restricted[20,26].

Many farms have at least one perennial or ephemeral waterway. Farmers may also create shallow drainage lines to assist in removing water (mainly during winter and early spring) from paddocks that are prone to waterlogging. Typically those paddocks are adjacent to waterways. The land alongside waterways on grazed dairy farms has usually been cleared of native vegetation with permanent pastures established, often right up to the stream bank. Farmers may also allow stock direct access to the waterway as a means of supplying the animals with water. Consequently riparian land has generally been managed for the purpose of dairy farm productivity with less emphasis on the detrimental impacts on the broader environment.

Threats to Riparian Ecosystems from Grazing

The riparian zone as defined in this paper is the 'area of land that adjoins, regularly influences or is influenced by the waterway'[27]. This definition does not exclude transient waterways, is based on proximity to surface and/or subsurface water and acknowledges the terrestrial—aquatic interactions in these zones[28,29]. The configuration of riparian landscapes is made up of longitudinal (i.e., corridors), transverse or lateral (terrestrial/aquatic links), vertical (links to groundwater and vegetation canopy) and internal structures. These features, stream size and its location in the drainage network (i.e., headwaters versus floodplain), as well as geomorphology, determine the shape, width and functions of riparian zones[28–30]. Riparian zones are also influenced by vegetation outside of this zone that contributes nutrients, organic matter and shade. On many dairy farms, the areas that farmers are willing to consider for riparian management are usually those most proximal to the stream, and therefore only include a portion of the ecological definition of riparian zones.

In undisturbed landscapes riparian zones support a range of aquatic and terrestrial organisms whose survival is interdependent and who play an important role in the cycling of nutrients in riparian zones[31,32]. Riparian ecosystems therefore also play an important role in maintaining and facilitating dispersal of biodiversity[28,30].

In disturbed environments, the structure, functions and processes in intact riparian zones can attenuate upland anthropomorphic impacts on waterways and the environment. For example, transverse structures physically separate land use impacts from the waterway. Where riparian zones are incorporated into upland land management, ecosystem processes (nutrient transformations, reductions in the velocity of moving water and sediment trapping) specific to undisturbed riparian zones will be modified due to changes in soil physical, chemical and biological properties as well as changes and loss of vegetation[33]. In these instances, a riparian zone may be less able to mitigate environmental impacts due to land-use activities. In fact, riparian structures, such as the vertical links to groundwater, may instead exacerbate environmental degradation. Additionally, loss of vegetation reduces the input of debris to supply nutrients, habitat and the geomorphic structures that moderate flow, decreasing aquatic and terrestrial biodiversity at local and catchment scales.

Poor riparian condition and in-stream water quality, as well as low or no native riparian biodiversity, are features of many grazed dairy farms[34,35]. Farm management activities on grazed dairy farms can be divided into those that have a direct or an indirect impact on riparian zones and waterways (Fig. 2). Activities that directly threaten the riparian zone include stock accessing waterways and riparian areas, and applications of fertilizers and effluent. Indirect impacts generally occur upslope and out of the immediate vicinity of the riparian zone and may be associated with normal farm management practices. Pollution from the directly and indirectly impacting activities can be further categorized into those that originate from point or non-point (diffuse) sources, with point sources of pollution usually most easily contained[7,36,37].

Indirect threats

Hoof impact. Movement of dairy cows on upland pastures can deform the landscape, creating trails, terracettes and enhanced surface roughness by compacting, shearing and moving, and smearing soils. A walking dairy cow could exert vertical pressures of at least 250 kPa[38]—more than double that when stationary[39,40]—and these are intensified during movement upslope, as can occur on steep riverbanks. In upland areas movement impacts can be significant as these soils are often shallower[38]. Decreased infiltration capacity in compacted and pugged (poached) soils results in increased and possibly changed runoff. Mesofauna, such as earthworms, are also disturbed by grazing, leading to decreases in hydraulic

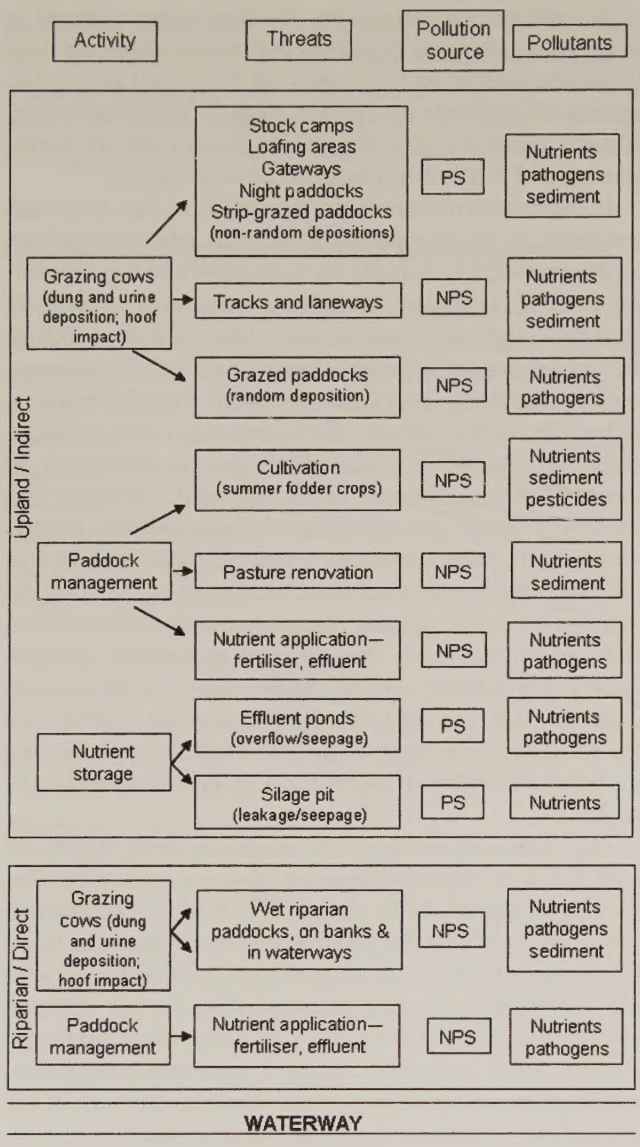

Figure 2. Threats posed by grazing systems on riparian zones and waterways from directly and indirectly impacting management activities, showing the pollution sources (PS, point source, NPS, non-point source) and the contaminants.

conductivity, soil structure and fertility, reducing the potential for recovery of compacted soils[39,40]. Moreover, the impacts of livestock on soil physical properties and pasture are not likely to be distributed evenly across a grazed paddock as cattle spend almost half of their time in less than 10% of the paddock[10,40].

Random excretal deposition. Dairy cows excrete a large percentage of nutrient intake in dung and urine[41]. While research in confinement dairy operations has demonstrated the effect of excess dietary P and N contents in dairy diets on excretions[42,43], limited research has been undertaken in grazing systems. Aarons[44] found in temperate Australian systems that seasonal production and nutrient content of grazed pastures affected nutrient intakes and dung nutrient concentrations (Fig. 3). Nutrient contents in the pasture and dung were highest in spring when pasture growth was greatest, then declined in summer. The feeding of conserved spring silage (with a higher nutrient content than available pasture) in late summer resulted in increased dung P concentrations. While the greater use of supplements in more intensively managed grazed pasture may even out the temporal variation in feed availability observed in purely grass-based systems, an increase in the total amount of P and N excreted by the animals is likely to be associated with the greater feed intake[42,43,45].

When excreta is deposited by grazing cows, P, K and N can be applied to soil at rates (e.g., $240\,kg\,P\,ha^{-1}$, $780\,kg\,K\,ha^{-1}$ and $1000\,kg\,N\,ha^{-1}$, respectively) well in excess of typical fertilizer application rates[14,41,46]. Consequently greater soil P, K and N fertility was observed under cattle dung pads[46–49], and urine patches[50].

Seasonal variations in the decomposition rate of deposited dung in the field could affect movement of nutrients from the pad and subsequent nutrient accumulation in soil[51]. In addition to nutrient loss from surface deposits of dung, nutrient movement in surface and subsurface water pathways from high-nutrient-content soils is likely to be a feature of grazing systems, after the dung pad has degraded[36]. Elevated soil P levels were observed for up to 112 days, although the dung pads had completely disappeared in approximately 60 days[52].

Nitrogen leaching losses in grazing systems are primarily determined by urine depositions[53], although greater nitrate leaching also occurs as N fertilizer application rates increase[54–56]. The impact of N fertilizer is two-fold; due in the first instance to the potential for fertilizer N to be mobilized, and secondly through promoting plant biomass and increasing pasture protein content[42]. Thus, while a linear relationship between fecal N and protein intake ($R^2 = 0.53$) was observed, urinary N increased exponentially ($R^2 = 0.95$), such that above an intake of $400\,g\,N\,day^{-1}$ greater amounts of N were excreted in urine compared with feces.

High concentrations of microbial pathogens are also deposited in dung and can include fecal streptococci, *Escherichia coli, Salmonella* spp., *Cryptospordium parvum* and *Giardia*, all of which pose a serious threat to human and animal health, and have been shown to be elevated in streams running through grazed pastures[37,57]. *E. coli* counts in dung of up to 10^6 most probable number (MPN) $100\,ml^{-1}$ or 10^9 cells per dung pat and *C. parvum* oocyte concentration of approximately $10^4\,g^{-1}$ feces have been reported, with both transported in runoff[26,58–60]. The numbers of *C. parvum* oocytes in runoff depended on the slope, presence of vegetation and on rainfall intensity. Despite the assertion by McNeill[61] that bacterial pathogen survival diminishes once feces are deposited by the animal, neither McDowell et al.[26] nor Muirhead et al.[60] observed first-order decay kinetics of *E. coli* in pads after deposition. Survival of pathogens in dung deposited for some time on pasture was thought to contribute

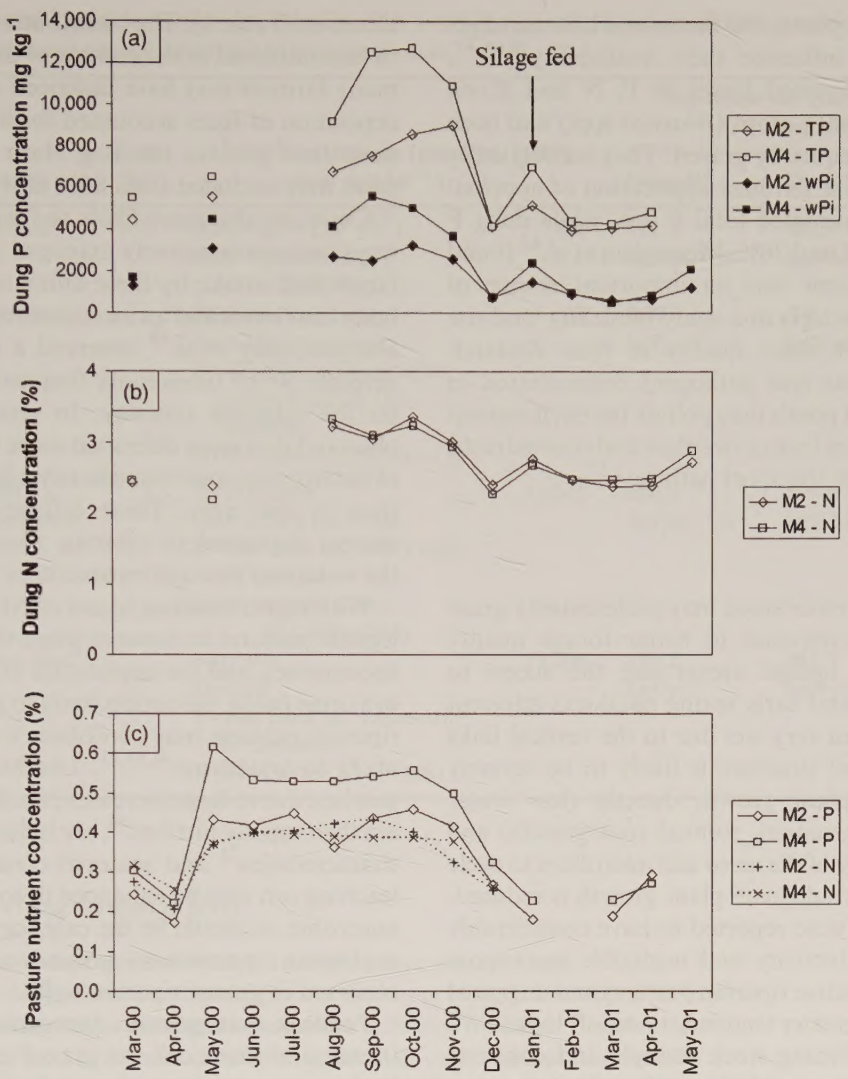

Figure 3. Changes in (a) dung P (Total P [TP], $mg\,kg^{-1}$; water extractable inorganic P [wPi], $mg\,l^{-1}$), (b) dung N (%) and (c) pasture P and N concentrations (%) over the 2000/2001 lactation, from cows grazing pastures that had been stocked at 3 cows ha^{-1} and received P fertilizer at 35 kg (M2) and 140 kg (M4) $ha^{-1}\,yr^{-1}$ for 6 years (Aarons[44]). No nitrogen fertilizers were applied over the 6 years of the study. Dung samples were randomly collected from 3 cows on each of 5 mornings once each month. Pasture samples were collected from paddocks that cows had grazed, except in February 2001 (summer) when there was insufficient pasture.

approximately 16% of the catchment *E. coli* load via overland flow[62].

Non-random excretal deposition. Animal behavior, driven by the availability and location of water and shade, landscape topographical features (e.g., hillslopes), and pasture growth and season, contributes to the concentration of animal excreta within the landscape and large-scale spatial heterogeneity in soil nutrient status[10,15,63]. Additionally, soil compaction and the resulting influence on soil hydrology will exacerbate the effect of stock camping behavior on movement of nutrients via overland flow[38]. Nutrient accumulation zones identified in the stand-off area near to the dairy shed appeared to cause elevated soil P levels down-slope and close to the waterway. Groundwater nutrient concentrations were affected in the areas with elevated soil nutrient levels[64].

Herd management (i.e., frequency of visits, duration and cow density) also influences spatial deposition of dung and urine and nutrient accumulation[22,44,64]. The use of night/day paddocks appear to lead to nutrient accumulation while uneven nutrient distribution was reported in strip-grazed paddocks where back fencing was not used.

Paddock management and nutrient storage. A number of studies have demonstrated the impact of fertilizer, dairy shed effluent and cultivation on nutrient, pathogens and sediment contamination of waterways when hydrological conditions are appropriate. For example, P and N runoff losses were related to time since grazing and fertilizer application, reflecting the diverse forms and sources of

these nutrients in soil, plant, and faeces and how management activities can influence their availability[26,65–67]. McDowell et al.[26] reported losses of P, N and *E.coli* from soil in which forage crops (*Brassica* spp.) had been grown and then subsequently grazed. They subsequently estimated P losses from fertilizer application of approximately 12% of the estimated total P lost, while dung P comprised between 25 and 36%. Monaghan et al.[62] found that dairy shed effluent was an important source of nutrients and fecal bacteria in a study modeling land-use impact on catchment water quality in New Zealand. Additionally, nutrients and pathogens concentrated in silage pits and effluent ponds may pollute the environment if these storage facilities leak or overflow and concentrated material reaches water transport pathways[37].

Direct threats

Hoof impact. In summer stock may preferentially graze riparian pastures in response to better forage quality (compared to more upland areas) and the access to water[15,38]. In winter and early spring paddocks adjacent to waterways are often very wet due to the vertical links in riparian zones. Soil structure is likely to be severely degraded affecting plant growth directly (low water infiltration, low oxygenation, stunted root growth) and indirectly as the ability of the meso and microflora to cycle nutrients required for adequate plant growth is reduced. Grazed pasture soils were reported to have considerably lower hydraulic conductivity and negligible macropore flow compared with native riparian pastures and ungrazed forests, suggesting a greater tendency to runoff losses[33,38].

Vegetation loss. Grazing stock trample and consume native riparian vegetation, with the resultant loss of understorey species increasing the likelihood of soil erosion, sedimentation of waterways, reduced aquatic life as well as declining native biodiversity such as woodland bird populations[68,69]. In contrast, Trimble and Mendel[38] reported that the greater grass growth observed when understorey vegetation was removed by grazing cows might decrease erosive losses.

An additional impact in Australian systems is the increased salinity observed in response to the replacement of native trees with annual and temperate pasture grasses[70], a likely outcome in grazed dairy pastures. The converse is also true, i.e., afforestation of a riparian zone can lower the water table and lead to lower stream flow, although the percentage decline (55–84%) varies in specific instances[71].

Excretal deposition. Deposition of feces and urine in waterways may directly introduce substantial amounts of nutrients and pathogenic organisms[58,72]. Presuming cows deposit dung an average of 12 times each day[41], each animal spends approximately 15 min standing in the stream, and that the riparian paddock is visited 17 times, a herd of 300 cows is estimated to deposit 0.8 and 3.4 kg P and N and 6.4×10^{11} *E. coli* cells, respectively, over a lactation (Table 1). The contribution from this herd needs to be considered in the context of dairy catchments where many farmers may have unfenced riparian zones. Direct deposition of feces accounted for 8×10^8 colony forming units $ha^{-1} yr^{-1}$ in the Bog Burn catchment, although cows were excluded from 84% of waterways[62].

Cows are also more likely to frequent unfenced waterways in more intensively managed grazing systems as the larger feed intake by these animals will result in greater body heat levels and an increased requirement for water[15]. Davies-Colley et al.[58] observed a tendency for cows to defecate 50–60 times more frequently in the stream than on the adjacent laneway. In contrast, James et al.[72] observed that cows defecated more frequently in the 10 m of the riparian zone immediately adjacent to the waterway than in the river. These differences notwithstanding, excreta deposited in riparian zones can also influence the waterway through transverse or vertical flow of water.

With higher leaching losses of NO_3 – N reported under legume pastures in summer when the legume component decomposes, and the application of up to 1200 kg N ha^{-1} in a urine patch, the connectivity to groundwater in grazed riparian pastures may contribute to increased movement of N to waterways[41,53,73]. Leaching losses from urine patches, due to large increases in soil solution NH_4^+, which are then rapidly nitrified[50], are influenced by soil drainage characteristics[73] and seasonal conditions[53]. Phosphorus leaching can also be enhanced in soils that are seasonally anaerobic, as would be the case for most riparian soils[74], explaining increases in groundwater P concentrations observed in grazed riparian soils.

Paddock management. Inaccurate application of nutrients and chemicals to grazed riparian pastures can lead to contamination of waterways with fertilizers, dairy shed and dairy factory effluent, and pesticides[37,75], while inadequate management of storage ponds (poor design, infrequent emptying) can result in loss of effluent to the waterway. Applying effluent to pastures, especially to riparian paddocks, can degrade water quality. Intensification of the industry resulting in increases both in the volume of dairy shed effluent, and in the concentration of nutrients in the effluent[75] is likely to increase this impact.

Shallow subsurface drains, installed to remove water from frequently flooded or wet riparian pastures, can provide a route for applied effluent to waterways. For example, Monaghan et al.[62] found that surface drains contributed 6% of the annual *E. coli* load, while mole-pipe drainage of effluent-irrigated pastures contributed 18% of the catchment P load[76].

Approaches to Improve Riparian Management

Traditionally, decreasing the degrading impacts of grazing systems on waterways has been based on the creation of a fenced vegetated buffer immediately adjacent to the

Table 1. Nutrient (N, P) and *E. coli* deposition rate calculated for areas of a grazed dairy farm receiving random or non-random dung deposits. Calculations are based on published and experimental data.

Deposition type	Location	Area (ha)	Annual herd deposits (potential, HDP) (cow visit $ha^{-1} yr^{-1}$)		Deposition rate (contaminant deposited in dung cow^{-1}*HDP)		
					Dung-P[6]	Dung-N	Dung-*E. coli*
– random	Paddock[1]	3.6[2]	HDP = (ρ[3] × Dn[4] × F[5]) HDP = ((300/3.636) × (128/24) × 17)	**7481**	**111.3** kgP $ha^{-1} yr^{-1}$	**482.9** kgN $ha^{-1} yr^{-1}$	**8.98×10^{13}** cells $ha^{-1} yr^{-1}$
– non-random[7] (influenced by animal behaviour)	Waterway		300 cows visit creek Dn = 15 min in creek *F* = 17 visits to this riparian paddock during the lactation HDP = (300 × (0.25/24) × 17)	**53**	**0.79** kgP yr^{-1}	**3.4** kgN yr^{-1}	**6.38×10^{11}** cells yr^{-1}
– non-random[8] (influenced by grazing management)	Night paddocks	3.6	*P* = 300 cows Dn = 12 h; 6 pm–6 am *F* = 304 visits i.e., every night for lactation HDP = ((300/3.636) × (12/24) × 304)	**12,541**	**186.6** kgP $ha^{-1} yr^{-1}$	**809.6** kgN $ha^{-1} yr^{-1}$	**1.50×10^{14}** cells $ha^{-1} yr^{-1}$
	Dairy shed[9]		300 cows Dn = 2 h *F* = 608 visits i.e., twice daily for lactation HDP = (300 × (2/24) × 608)	**15,200**	**226.2** kgP yr^{-1}	**981.3** kgN yr^{-1}	**1.82×10^{14}** cells yr^{-1}

Farm data
Herd size: 300 cows.
Grazing area: 200 ha.
Lactation length: 304 days (Aug. through to May).
Milking times: 6–8 am; 4–6 pm.
Rotation lengths: Spring—15 days; summer 30 days; autumn 21 days; winter 30 days.
- Spring—cows graze during the day and overnight—then return in 15 days, from Sept. 1 to Nov. 30.
- Summer—cows graze for 2 days and nights—then return in 30 days from Dec. 1 to Mar. 31.
- Autumn—cows graze for 2 days and 1 night—then return in 21 days from Apr. 1 to May 31.
- Winter—cows graze for 2 days and nights—then return in 30 days from Jun. 1 to Aug. 31.

Number of paddocks: 55 paddocks.

[1] Refers to where cows graze as part of their lactation. Note: not including night paddocks.

[2] Average area = 200 ha over 55 paddocks.

[3] ρ = Number of cows ha^{-1}; based on the size of the location.

[4] Dn = Number of hours for each visit on a per day basis; Dn_{sp} = 8 (day) + 12 (night) = 20; Dn_{su} = 16 (day) + 24 (night) = 40; Dn_a = 16 (day) +12 (night) = 28; Dn_w = 16 (day) + 24 (night) = 40.

[5] *F* = Number of visits for lactation.
F_{sp} = 7 visits; F_{su} = 4 visits; F_a = 3 visits; F_w = 3 visits.

Assumptions
Dung deposition cow^{-1} day^{-1}: 12 pads (James et al.)[72].
Average dung dry wt: 0.2 kg pad^{-1} (Haynes and Williams[41], Aarons[44]).
Average dung P, N concentration: 0.62, 2.69% respectively (Aarons[44]; see Fig. 3).
Average *E. coli* concentration: 10^9 cells dung pat^{-1}.

[6] *Calculations of contaminant deposition per cow*
Total P dung $deposit^{-1}$ = 0.2 × 0.62% = 0.00124 kg.
Total P deposited cow^{-1} day^{-1} = 0.2 × 12 × 0.62% = 0.01488 kg.
Total N dung $deposit^{-1}$ = 0.2 × 2.69% = 0.00538 kg.
Total N deposited cow^{-1} day^{-1} = 0.2 × 12 × 2.69% = 0.06456 kg.
Total *E. coli* cells $deposit^{-1}$ = 10^9 cells.
Total *E. coli* deposited cow^{-1} day^{-1} = $10^9 \times 12 = 1.2 \times 10^{10}$.

[7] Non-random deposits influenced by animal behavior also occur in camp areas (e.g., water points, shade/shelter, topographical features in the landscape), laneways and feedpads. Contaminant deposition in these areas has not been calculated.

[8] Non-random deposits influenced by grazing management also include strip-grazing and conservation paddocks. Contaminant deposition in these areas has not been calculated.

[9] If the area is known of the part of the dairy shed where excreta is deposited and collected, then loading rates on an area basis can be calculated.

waterway. However, changed management both of the riparian zone and upland areas is recommended, with changes to typical day-to-day farm activities required to augment the effectiveness of established buffer strips in attenuating losses from fields[37,77,78]. The effectiveness of changed management depends on the extent of direct and indirect land-use impacts, how effectively these impacts can be moderated and how much of the riparian zone is appropriately managed.

Upland management

Diminishing losses from non-random excretal deposition may require infrastructure to contain and improve the collection and storage of excreta from feed pads and stand-off areas[79], changing paddock grazing management to restrict the time cows graze cropland or pasture in winter, and providing alternate water sources and managing forages to control where cows camp[15,18,62]. Targeted applications of fertilizer or effluent at times when runoff is less likely to occur, based on soil testing[80,81] and considering the solubility of the fertilizers used[18,62], will minimize non-point sources of nutrients. Likewise, conservation tillage practices can reduce sediment and particulate nutrient losses[82]. However, the impacts from random deposition of dung on grazed paddocks are less easily managed through improved farm practice.

Riparian management

Within riparian areas practices similar to those for upland areas apply[79–81]. However, there are little data in the literature supporting the on-site and catchment benefits to be accrued from implementing grazing best management practices. In one such study McDowell et al.[26] suggest that total P losses could decrease by one-third when animals are moved away from riparian areas, based on dung P comprising between 25 and 36% of estimated total P losses. However, Agouridis et al.[15] recommend critical investigation of grazing best management practices that are expected to deliver enhanced water quality.

Buffers. Riparian buffers (or variously named, riparian forest buffers, vegetated buffer strips, grass filter strips, vegetated filter strips, etc.) offer opportunities to reduce nutrient and pathogen losses to waterways while improving biodiversity values. However, fencing, buffer widths, species selection and the configuration and management of these strips are design issues that can greatly influence how effectively riparian buffers mitigate the impacts of grazed dairy systems[77,83,84].

Fencing. In addition to improvements in water quality[62,79,85], fenced revegetated riparian zones had significantly greater native small mammals, birds and vegetation compared with unfenced grazed sites[34]. However, the lost production associated with fencing riparian buffer areas[15] as well as fencing costs are often considered a substantial disincentive for many farmers[86]. In the associated study riparian pasture production was, on average, 25% greater than that on more elevated parts of three dairy farms and was considered a valuable feed source by farmers[86]. However, the fenced pasture represented only 0.2–1.7% of that available for milk production. These farmers indicated that fencing riparian areas improved herd management; specifically less time spent locating stock. Improvements in animal health and production from minimizing access to contaminated water are also likely. Despite the production and management benefits, how readily farmers fence and revegetate riparian zones often depends on the width of the zone to be managed.

Widths. The width of the fenced zone will greatly influence the effectiveness of the riparian buffer; however, there is little consensus in the literature as to the optimum width. Widths of as little as a meter were suggested to be of some benefit to the environment, trapping sediment as well as soluble nutrients if infiltration is increased[77]. However, Collins et al.[87] recommended buffer widths of greater than 5 metres to increase the entrapment of *E. coli* and *Campylobacter* under high flow rates. In contrast, McNeill[61] suggested that although buffers up to 30 m decreased bacterial concentrations in runoff, they were not adequate to meet water quality guidelines.

Grass buffers were reported to retain most sediment in the first 5 m, while over 15 m was required to decrease bioavailable P by more than 60%[78]. These authors recommend basing riparian grass buffer widths on the ratio of the buffer strip area to the source area, especially where topography concentrates water flow toward the buffer zone. While results in the literature are variable, they generally indicate that high ratios are better suited to attenuating nutrient loss from arable (i.e., cultivated) land especially in high-flow events. However, concentration of water flow through buffers can mean that the effective buffer area is considerably smaller than the gross buffer area, substantially reducing the sediment trapping capacity of the buffer[88]. In addition, this effectiveness is further reduced by soil compaction[78].

Narrow-fenced buffers are also less suitable from an ecological point of view as fencing recommendations for the creation of wildlife corridors are invariably much greater[89]. Recommended widths for most birds (including large birds of prey) and animals are 50–100 m[90,91]. Price et al.[91], recognizing the impracticality of these widths, suggest islands of 50–80 m with 20 m corridors in the intervening space.

Vegetation. Forested riparian buffers potentially create the most variable and hence ecologically favorable habitats. Consequently, restoration of riparian forests is frequently preferred from a biodiversity point of view[92], although Osbourne and Kovacic[84] suggest that grass riparian buffers also fulfill this function in the mid-Western United States. Forested riparian buffer zones generally increase water infiltration, reduce sediment loss from the surrounding land and promote sediment deposition and nutrient retention[84,93–98]. For example,

average sediment loss declined by an order of magnitude after establishment of a forested riparian buffer. Phosphorus and N loss in runoff from agricultural land can also be significantly reduced; N more so than P. Groundwater nitrate is removed either through plant uptake via tree roots or denitrification processes in the soil although this was dependent on the depth to the water table[84,94,99]. In addition, the forested vegetation contributes terrestrial and aquatic (woody debris and snags) habitat, patch connectivity and reduces dieback of remnant vegetation[15,28,100].

Despite the benefits mentioned above, the presence of a riparian forest buffer is not always beneficial to water quality. Total P export remained the same after fencing and establishment of a *Eucalyptus* forest buffer but the proportion of reactive P increased[96]. Osbourne and Kovacic[84] reported increased total and dissolved P concentrations in shallow lysimeters below riparian forests compared with crop land where these were located immediately adjacent to a waterway. Forested riparian buffers can also result in reduced ground and lower storey vegetation as the tree canopy closes[38,101] leading to greater surface water movement through the riparian zone[71,102]. However, the improved hydraulic conductivity of forested riparian soils may compensate for the greater overland flow[33]. Reduced water yield, due to increased transpiration by trees may also decrease the extent of wetlands as well as in-stream vegetation contributing to higher stream baseflow nitrate concentrations[71], although groundwater nitrate declined under trees[84].

The tendency of cows to seek shade and shelter and to graze near to waterways may further decrease the effectiveness of forested riparian buffers. Contaminant transport could readily occur in conditions of infiltration-excess or saturation-excess overland flow[103], while shallow groundwater would provide a translocation pathway for nitrate-N from urine and dung. Areas of bare ground such as gateways, water troughs and feeding areas may position high contaminant areas within a few meters of a forested buffer which may be incapable of attenuating the pollutant threat to the waterway. Monaghan et al.[104] observed that although areas used for dairy wintering occupied only 10% of the catchment area, they contributed up to 60% of modeled N losses. Greater concentrations and loads of *Cryptosporidium* oocytes were recovered in runoff from fecal pads created on bare soil blocks compared with blocks with more than 90% vegetated cover[59], particularly during low-intensity rainfall events of long duration. The distance the oocytes moved depended on topography and vegetative cover. While bacterial loads declined by 95% when fecal deposits were located a minimum of 2.5 m from the waterway, this could still result in 5×10^4 MPN $100\,ml^{-1}$ reaching the waterway in runoff from each fecal deposit[15,60].

An alternative to riparian forest buffers is grass buffer strips, which have been shown to significantly reduce nutrient and sediment losses[105,106] through increased infiltration, sediment deposition and filtration processes[83,107]. For example, the continual cover and growth afforded by perennial grasses reduced runoff volume and velocity through absorption of up to 90% of the shear stress[78]. The grass cover also decreases raindrop impact with a minimum of 70% cover recommended to control hillslope erosion and dense ground cover required to prevent gully erosion, especially at degraded sites. Adsorption of colloidal particles to grass is likely to contribute to very minor nutrient trapping while hillslope and gully erosion control reduces loss of organically bound N and P as well as particulate P[83].

As with forest buffers, denitrification and/or plant uptake can slow nutrient loss in grass buffer strips. These processes require long retention times within the grassy buffer. However, it is important to note that the effectiveness of a grass buffer depends on its vegetative structure, width, soil type, position in the landscape and management, and will vary from time to time[78,89]. For instance, N leaching is increased under pastures with a clover component[14,15], while P retention in a grass buffer can vary seasonally and may decline over the longer term. Additionally, resistance to flow decreases if vegetation is submerged or if grass height is too great leading to lodging and increased water movement via preferential routes[78].

The historical debate regarding the most appropriate vegetation for riparian zones[99,108] continues. Dosskey et al.[109] compared the effectiveness of grass and forest (half-trees and shrubs, half-grass) buffers and observed no difference between the two types in the first ten growing seasons, due primarily to the similarity in ground cover. The greatest improvement occurred in the infiltration capacity of the filters, while some improvement in sediment deposition and little or no difference in dilution processes were observed. Alternatively, grass buffers were more effective than *Eucalyptus* buffers at reducing nutrient and sediment movement from sheep-grazed pasture. No runoff was recorded in the grass buffer in summer, while the forested buffer functioned as a sediment source throughout the year[102,103]. Soils under the eucalypt buffers appeared to have developed a surface crust, due to ploughing and subsequent compaction, which lowered infiltration. In contrast, Cooper et al.[33] found that the saturated hydraulic conductivity of native forest and set-aside riparian areas was at least 6 times greater than that of grazed riparian land, most likely due to the natural recovery of soil physical properties when grazing animals are excluded[39].

The set-aside soils had larger pools of bioavailable P, although they had not received P fertilizer for 12 years. In a review of a number of studies, median total P declined by 15% under riparian forest buffers compared with decreases of 50% for grass filter strips, while median dissolved P losses were greater under forest filters (60%) than grass filter strips (30%)[80]. Osbourne and Kovacic[84]

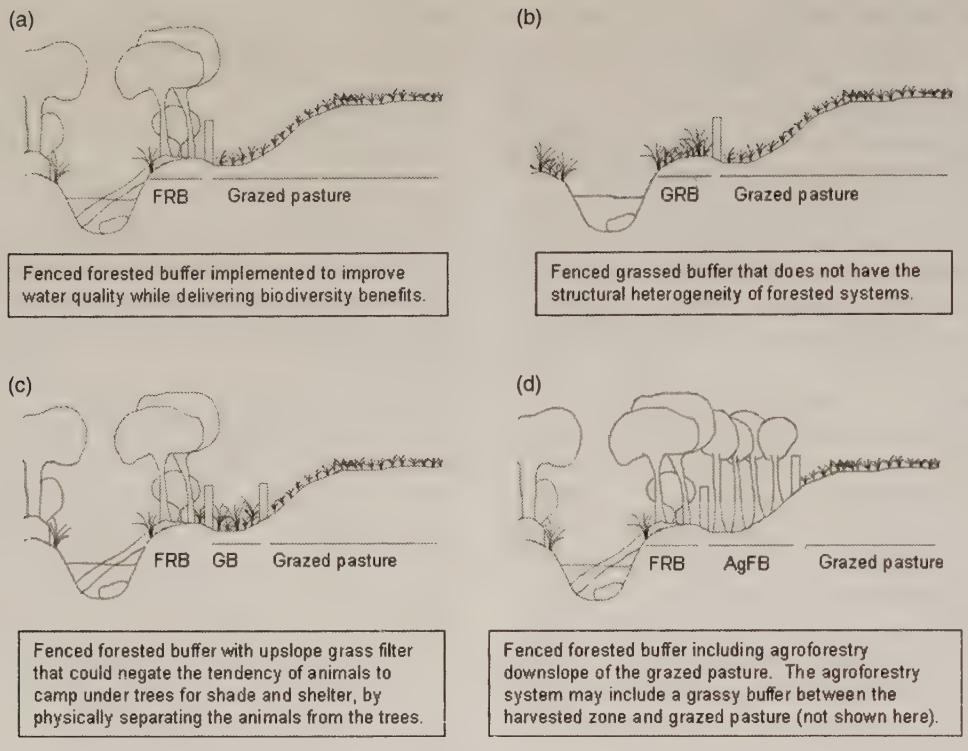
(a)
FRB
Grazed pasture
Fenced forested buffer implemented to improve water quality while delivering biodiversity benefits.
(b)
GRB
Grazed pasture
Fenced grassed buffer that does not have the structural heterogeneity of forested systems.
(c)
FRB
GB
Grazed pasture
Fenced forested buffer with upslope grass filter that could negate the tendency of animals to camp under trees for shade and shelter, by physically separating the animals from the trees.
(d)
FRB
AgFB
Grazed pasture
Fenced forested buffer including agroforestry downslope of the grazed pasture. The agroforestry system may include a grassy buffer between the harvested zone and grazed pasture (not shown here).

Figure 4. Different configurations of riparian buffer strips that can be used on grazed dairy farms to mitigate contamination of waterways. FRB, forested (native vegetation) riparian buffer; GRB, grassed riparian buffer; GB, grassed buffer (may be strategically grazed or mechanically harvested; AgFB, agroforestry (native or exotic) buffer.

reported that grass buffers more effectively reduced total and dissolved P losses, although the trees assimilated more nitrate-N.

Alternative riparian management and configuration

A well-managed pasture that is not overgrazed could constitute a grass filter strip. Strategic grazing of grass buffer zones[110] at lower stock intensities during the drier months of the year would contribute carbon inputs in feces that may sustain long-term denitrification requirements of these areas[39,40,99], although stock access should be restricted during the wetter months. Lower fertilizer use in the grassy buffer strips and incorporation of native grasses can contribute to increased biodiversity values, although native grasses do benefit from small additions of fertilizer (M. Mitchell, pers. comm.). An alternative to strategic grazing is to use the grass buffer strips for fodder conservation and withhold grazing from these areas, particularly as mechanical harvesting contributes less soil physical damage than grazing[39]. Buffers such as these can be placed downslope of summer fodder crops or reseeded pastures[77].

An alternative riparian configuration is the combination of a grassed buffer strip managed alongside and upslope of a forested riparian buffer (Fig. 4). A three-zone riparian forest buffer system incorporating an agroforestry zone is the recommended approach for establishing riparian buffers in the USA[77,111]. The forests are managed as a source of additional on-farm income with the advantage of increasing the area revegetated alongside rivers and streams. Water quality was not compromised by harvesting timbers within this 45–55 m zone (zone 2) located between an 8 m grass filter strip upslope (zone 3) and a 10 m permanently forested zone downslope (zone 1) and adjacent to the river. Grass buffers in conjunction with forest filter strips should eliminate virtually all sediment and associated nutrient loss, as these strips can each trap in excess of 90% of sediment loss from agricultural sources[107].

Naiman and Décamps[30], as well as researchers at Iowa State University and the University of Missouri Center for Agroforestry[112], also describe 'multi-species' riparian buffers consisting of three zones (albeit of different widths) where the function of the two zones closest to the river is to maximize contaminant removal, while the third zone slows water movement, thereby contributing to coarse sediment deposition. Infrequent harvest of the second zone improves the functions of zone 1, suggesting that appropriately managed agroforestry would improve environmental outcomes.

Despite the potential economic benefit and subsequent incentive for farmers to implement a zoned approach to improving riparian management, current agroforestry codes of practice discourage the harvesting of riparian

forest buffers in Brazil and many states in Australia. The converse is true for Germany and the USA where many farmers adopt riparian forest buffers for both environmental and economic benefit[113,114]. Water quality concerns associated with harvesting (i.e., logging) appear to influence theses codes of practice for riparian forestry. Dominant sediment sources, the general harvesting area and buffer widths are important management factors for protecting waterways from logging operations[115,116]. However, a number of planning tools for determining optimum widths and placement of riparian buffers have been developed[117–119].

An additional riparian buffer configuration is the installation of wetlands to manage nutrient losses in cases where subsurface flow dominates transport processes, as recommend by Osbourne and Kovacic[84]. Wetlands should be designed to maximize retention times based on the contaminant of concern and can be very effective in reducing nutrient losses to waterways[93,120]. This approach was adopted downslope of cropping in the riparian management system, but apparently not considered for grazed pastures[112], despite nutrient losses in drainage from grazed pastures[18,62,76].

Future Research Needs

Decreasing the environmental impact of intensive grazed dairy systems requires quantification of the nutrient inputs, flows and inefficiencies in these systems, and integration of these with hillslope–riparian–stream source transport models for N and P to better manage upland nutrient sources[18,22,26,62,66]. Improving the efficiency of nutrient use by dairy cows is an important first step[121], particularly the relationship between nutrient intake and excretion in grazing systems, where pasture intake can be less accurately defined. Dietary intake may then be manipulated to reduce nutrient loading rates. When linked to the distribution of animals throughout the landscape, the extent of nutrient loss to the environment from diffuse or point sources can be more accurately quantified and better managed.

While fencing waterways to exclude stock is important the width, vegetation types and configuration of fenced zones in grazed systems need to be examined to ensure improved catchment outcomes. The suitability of the buffer area/source zone ratio should be tested as one means of determining appropriate widths for grazing systems[78]. Ideally the effectiveness of these ratios should be measured downslope of grazed pastures, grazed forage cropland, and nutrient accumulation areas such as waterpoints or gateways. Vegetation types for fenced riparian zones to meet ecological and water quality requirements, and the suitability of multi-zone riparian buffer systems that incorporate a grassy filter strip with a forested buffer need to be evaluated. A configuration to be tested in grazing systems could also include the three-zone system recommended by the NRC for cropping systems[77,112] and incorporate wetlands to manage subsurface flow in drains[121].

Whichever configuration is identified as most suitable, both the short- and long-term effectiveness of the selected riparian buffer will need to be assessed, particularly using widths typical of those used by farmers[89]. These assessments will give farmers and natural resource managers confidence that recommendations will not contribute to environmental degradation in the future. One approach is to compare sites that are grazed, have been fenced or are undisturbed to measure differences due to changed management over time[33,34]. An alternative is to establish grass and forested buffers and measure their effectiveness at attenuating sediment and nutrient loss over time compared with established grass or cropland[109]. The latter approach has the disadvantage of requiring a longer study period for the experimental design. In addition, year-to-year variation may confound results. For example, Dosskey et al.[109] could not explain the unexpected nitrate and nitrite–N enrichment observed in the first season of a study investigating the effectiveness of filter strips in mitigating surface runoff losses. In contrast, taking measurements at different sites that have previously had fencing/vegetation treatments imposed has the benefit of more rapid assessment, although confounding location differences will have to be carefully managed. Rainfall simulators are often used to facilitate runoff measurement. However, the higher rainfall intensities used often result in infiltration-excess overland flow, rather than saturation-excess overland flow, influencing runoff sediment and nutrient concentrations and the conclusions that can be drawn for pasture systems[66,102,103].

Monitoring riparian soil P status, including the P sorption characteristics of different soil horizons is recommended as a means of determining when fenced riparian strips are approaching P saturation, and therefore the possibility of releasing P to the environment[74,122]. This method should be tested on a range of soils in fenced riparian zones to determine the potential for P saturation of these areas. The effectiveness of grazing or harvesting to mine soil P, or mixing topsoils with subsoils needs to be evaluated, especially those using dairy cows for P removal.

Conclusions

Grazing systems are largely defined by cow movement in the landscape. The density of animals, frequency and duration of visits are a function of animal behavior and grazing management practices, which influence random and non-random deposition of excreta around the farm. Consequently, grazed dairy systems are likely to contribute to waterway and catchment degradation differently to confined dairy operations.

The preference of cows to camp near forested areas and the deposition of feces and urine with high nutrient and bacterial loads cast doubt on water quality improvements expected for current recommendations for fencing and revegetating riparian zones in grazed dairy systems. While forested and grassy buffers can potentially improve water quality, a combination configuration is recommended to meet ecological and biodiversity requirements while minimizing contaminant loss from grazed dairy farms and contributing to farm productivity. However, appropriately designed experiments are required to elucidate their effectiveness in the long term and in comparison with current recommendations. To have widespread applicability, these experiments need to take into consideration different topographies, runoff intensities and source areas (camp areas versus grazed pasture) and investigate sediment, nutrient and pathogen retention.

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The influence of organic transition strategy on chemical and biological soil tests

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Abstract

Soil testing strategies that include biologically based indicators in organic and alternative farming systems are needed in order to improve recommendations that balance production and environmental goals. In this study, soil samples were collected before and after soils were transitioned from conventional row crop production to organic management using rotations that varied in both their inputs and tillage intensity. Ley-, row crop- and vegetable-based farming systems were implemented using locally specific production practices. Subplots were imposed within each system to allow comparison of farming systems without amendment and with dairy manure- and compost-amendment. Soil analyses included standard chemical tests (0–15 cm) for available phosphorus, exchangeable potassium, calcium, magnesium, pH, total organic carbon (SOC) and total nitrogen (TN). Biological assays (0–15 and 15–30 cm) included particulate organic matter-C and -N (POM-C, POM-N), soil and POM C:N ratios, fluorescein diacetate (FDA) hydrolysis, potentially mineralizable N (PMN) and hydrolysable amino-N + NH_4 (IL-N). Even though cropping and tillage intensity varied among systems (ley < row crop < vegetable), organic matter and nutrient reserves were not statistically different. Nutrient concentrations tested medium to high, even without compost or manure application. Labile fractions of soil organic matter were more enriched in the deeper sampling depth; whereby, POM stocks within the 15–30 cm depth increased by 20% on average compared to roughly 6% in the surface depth. This and observed changes in other properties demonstrate the multiple benefits derived from use of winter annual or perennial crops. Results from our analyses suggested PMN and POM have particular promise as metrics of change in commercial soil testing facilities to assist recommendations for amendments to balance production and environmental goals.

Key words: soil testing, biological indicators, organic farming systems, organic transition, soil organic matter, labile soil organic matter, scoring functions

Introduction

Organic standards emphasize the maintenance or improvement of the soil resource by building soil organic matter and increasing soil biological activity[1]. Organic management practices are intended to satisfy plant nutritional needs without drawing down soil reserves or polluting adjacent air or water resources. Several studies comparing organic and conventional agriculture have suggested that use of organic practices increases soil organic matter, soil biological activity and plant available nutrient stocks[2–8]. Most comparisons have been of conventional-tilled cash-grain systems with more diverse organic rotations amended with compost or animal manures[6,9,10]. Less is known about the ability of organic vegetable-cropped systems or systems that are not amended with animal waste to build soil organic matter and increase fertility and biological activity. Results from horticultural systems have been mixed and appear to vary with disturbance intensity and amendments applied[11,12]. Systems relying exclusively on plant-based fertility[13] may deplete soils through nutrient removal and lack of input, particularly for P and K. Appropriate testing tools would help growers evaluate the status of their soils and tailor management to maintain soil quality and yield without causing environmental harm[14,15].

To make standard soil tests attractive to organic growers, approaches need to be modified. Standard testing protocols are used to determine whether available nutrient stocks will satisfy crop requirements with the goal of maximizing yield[16,17]. The 'basic soil test' typically measures pH, total soil organic matter, available P, exchangeable K, magnesium (Mg), calcium (Ca) and sulfur (S) in soils collected from the surface depth only

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(0–15 cm) and before planting. Additional services frequently include determination of micronutrients if soils are naturally deficient. Accepted measurement techniques vary by region, with soil type and cropping complexity[16]. For example, soil testing methods in Illinois have been adopted following recommendations by the North Central Regional Soil Testing Committee[18] and their interpretation is guided by recent scientific research and delivered through Extension outlets. To serve organic growers, the basic soil test could be broadened to include soil quality measures that have the potential to provide insight into soil functions regulated by organic matter and microbial activity[13]. Some of the most promising candidates for organic soil testing include labile fractions of soil organic matter and N availability[15,19], which are often enriched compared to conventionally managed controls[20–22]. Soil testing for organic systems might also be improved by sampling organic matter and nutrient stocks contained within the full management zone, which extends beyond the current norm for soil testing in the top 15 cm of soil profile.

To develop an interpretive framework to evaluate standard and biologically based indicators within organic systems, upper and lower boundaries related to environmental and production concerns are needed. By establishing clear upper boundaries we can help growers avoid situations where nutrient concentrations are in excess and could degrade adjacent resources, such as water[23], or have negative feedbacks on forage quality[24]. Relationships between multiple desired outcomes and soil test values can be established using non-linear scoring curves[25]. These types of curves (more is better, less-is-better and mid-point optimum) are commonly used for utility functions[26], multi-objective decision making[27] and systems engineering[28]. Efforts to develop and adapt these basic curve shapes for soil applications have relied on literature review with indicator selection being based on statistical reduction and expert opinion[29]. Functional relationships and associated weighting factors have been established using expert judgment based on indicator sensitivity to management, inherent soil properties and climate factors. The scoring curve for soil organic matter, for example, is an ascending logistic *S*-curve, or more-is-better function with a plateau, based on the contributions organic matter makes to fertility, water partitioning and structural stability[29]. It might be appropriate to modify this function and others to include optimum values and associated declines in function with further increases in organic matter to protect the environment and encourage efficiency. For example, organic matter levels are thought to reach a maximum or saturation point that represents a dynamic equilibrium that is soil and system dependent[30]. Setting appropriate maximum threshold values based on this understanding could prevent over-application of manures and composts and prevent unwanted waste or pollution. Similar ideas apply to organic fractions in soils, which cannot accumulate to unlimited levels without becoming problematic[31]. Reviews by Watson et al.[10] and Nelson and Janke[32] showed how both P and K increased to high levels in organic systems when surplus manure was applied. To avoid depletion or 'mining of nutrients' from soil (e.g., Andrist-Rangel et al.[33]), lower limits need to be established. Studies have shown that both soil P and K test levels decline in some organic cash grain systems[5,34], but it is not clear when and how those levels would limit productivity over time.

The goal of this study was to contribute to the development and application of soil tests for organic farming systems. The research was conducted in a study established at the University of Illinois at Urbana–Champaign with three different farming systems or strategies to transition to organic certification. Field plots managed under a perennial ley, row crop and vegetable system each received three types of organic amendments (cover crops + raw manure, cover crops + composted manure and cover crops only) applied at times and rates appropriate to the system. We expected soil resource status, estimated through conventional soil tests, to decline in the intensive vegetable and row crop-systems that received no manure or compost during transition. Additionally, we expected to detect soil test increases in systems with the lowest disturbance frequency and greatest amount of organic matter addition.

Our specific objectives were to evaluate: (1) changes in soil nutrient status and organic matter resulting from three distinct farming systems used in transition to organic production; (2) changes in soil properties expressed at two soil depths and (3) the commercial potential of biologically based test methods. Finally, we wanted to outline a framework for test interpretation and application to organic farming systems using a scoring curve approach. This method was selected because it was considered well suited to growers seeking to balance economic and environmental performance[24,35,36].

Materials and Methods

Site and treatment description

The Windsor Organic Research Trial occupies a 2.5 ha site on Catlin silt loam (moderately well drained, fine-silty, mixed, superactive, mesic Oxyaquic Argiudolls) and Drummer silty clay loam (poorly drained, fine silty, mixed, superactive, mesic Typic Endoaquolls) soils located at the University of Illinois at Urbana-Champaign campus. During the 4 years preceding the start of this study, the areas had been used for conventional soybean (*Glycine max* (L.) Merr.) production with inputs based on soil tests. In the spring of 2003, 36 experimental plots (33 × 13 m) were established to begin a 3-year transition period to meet the certification criteria for the National Organic Standards[1,37]. Treatments were arranged in a split-plot design randomized in four blocks, in which farming systems were randomized to whole plots

Table 1. Crop rotation and field management summary at the Windsor Organic Research Trial during the three-year transition period.

Farming system	Rotation	Amendment	Amendment incorporation	Tillage[1]
Ley	Alfalfa, red clover, timothy, orchardgrass, alsike clover	• Compost (NPK equivalent to 622, 504 and 352 kg ha^{-1}) • Manure (NPK equivalent to 780, 259 and 461 kg ha^{-1}) • Cover crops only	Spring 2004, Fall 2005 Spring 2004, Fall 2005	Moldboard plough (20 cm, May)[2], Disk (15 cm May), seeding pasture mix, 2 mowings for weed control—compost and manure application (June), 4 mowings for weed control—mowing for weed control, manure and compost application (November)
Row crop	Soybean-winter wheat-hairy vetch-corn, winter rye	• Compost (NPK equivalent to 578, 480 and 444 kg ha^{-1}) • Manure (NPK equivalent to 324, 115 and 217 kg ha^{-1}) • Cover crops only	Fall 2003, Fall 2005 Fall 2003, Fall 2005	Moldboard plough (20 cm, May), Disk (15 cm May), soybean planting, 2 passes of cultivator for weed control (10 cm, June and July), compost and manure application (October), seeding winter wheat (October)—wheat harvest and straw raked and removed (July), hairy vetch drilled (September)—mowing and incorporation of hairy vetch (15 cm, May), disk/harrow for seed bed preparation (15 cm, May), corn planting (June), cultivator for weed control (10 cm, June and July), manure and compost application (November)
Vegetable	Tomato, Austrian peas-broccoli, wheat/hairy vetch-winter squash, winter rye	• Compost (NPK equivalent to 324, 115 and 217 kg ha^{-1}) • Manure (NPK equivalent to 323, 114 and 189 kg ha^{-1}) • Cover crops only	Spring 2004, Fall 2005 Spring 2004, Fall 2005	Moldboard plough (20 cm, May), Disk (15 cm May), and seed bed preparation for tomato transplanting (May), disk/harrow for residue incorporation (15 cm, October), disking and Austrian winter pea seeding (15 cm, October)—mowing and incorporation of cover crops (10 cm, March), seed bed preparation and broccoli transplanting (15 cm, June), cultivation for weed control (10 cm, July), manual weeding, disk/harrow (15 cm, October), winter wheat/hairy vetch seeding (October)—disk/harrow for cover crop incorporation (15 cm, May), row preparation for squash transplanting (10 cm, May), cultivator for weed control (10 cm, June), manual weeding, manure and compost application (November), winter rye broadcast (November)

[1] Growing seasons are separated by em dashes.
[2] Indicates the depth and month of cultural practice.

and organic matter amendments were randomized to split plots. The experiment compared three transitional farming systems used by growers in the US Midwest, varying in their need for inputs and cash flows. These farming system treatments were selected after consultation with an advisory board comprised of Illinois growers who had been farming organically for a decade or more. The treatments were: (1) ley fallow (low intensity); (2) row crop (intermediate intensity) and (3) vegetable (high intensity) systems. Organic matter inputs were: (A) cover crops + composted dairy manure; (B) cover crops + raw dairy manure, and (C) cover crops only. Each split plot was 372 m^2. Applications of organic amendments were made on a system basis to satisfy N requirements, and timed to reflect practices commonly used by producers. Details on management practices are summarized in Table 1. Inclusion of a 'conventional control' that continued the previous practice for purposes of comparison

was not appropriate for an organic transition study for several reasons, including restrictions by the granting agency and certification requirements that prohibit exposure to agrichemicals.

The entire experimental area was disked once and winter rye (*Secale cereale* L.) was seeded in the fall of 2002 before treatments were initiated in the spring of 2003. At that time, the plots assigned to the ley system were planted with a hay and pasture mix (untreated non-organic): alfalfa (*Medicago sativa* L. 'Big Sky'; 29.5%), red clover (*Trifolium pratense* L. 'Arlington'; 19.9%), timothy (*Phleum pratense* L. 'Climax'; 19.9%), orchard-grass (*Dactylis glomerata* var. 'Potomac', 19.5%) and Alsike clover (*Trifolium hybridum* L.; 9.9%). This mixture was consistent with the transition and rotation strategies used by our collaborating farmers who were not limited by land availability. To prevent the establishment of perennial weeds, these plots were mowed to 15 cm two times in 2003, four times in 2004 and once in 2005 and all residues were left in place. A similar effect would have been achieved had we harvested the forage as hay. The row crop system was seeded to soybeans, followed by winter wheat (*Triticum aestivum* L., Gleason Grains, Bridport, Vermont) in 2003. When wheat was harvested in the following year (2004), straw was baled and removed from the field plots. After a brief fallow period, plots were planted with a hairy vetch (*Vicia villosa* Roth) green manure cover crop, which was followed in 2005 with corn (*Zea mays* L.) and winter rye. The vegetable system was planted in 2003 with tomato (*Lycopersicon esculentum* Miller) on raised beds. After harvest crop residues were incorporated and no winter cover crop was used. In the late winter of 2004 plots were disked and seeded with Austrian winter peas (*Pisum sativum* spp. *arvense* (L.) Poir.) as a green manure that was followed by broccoli (*Brassica oleraceae* L. 'Gypsy'). After harvest, plant residues were flail mowed and incorporated before planting a mixture of winter wheat/hairy vetch cover crop followed by winter squash (*Cucurbita maxima* Duchesne ssp. *andreana* (Naudin) Filov) in the last year of transition. After harvest, winter rye cover crop was seeded.

Soil sampling and analyses

Soil samples were taken using soil testing protocols that are common in the North Central region. Soil samples taken in Spring 2003 before treatments were established were used as the baseline against which post-transition samples taken in Spring 2006 were compared to evaluate whether use of organic practices improved soil quality as is required by the National Organic Program standards[1]. To avoid edge effects, soil samples were collected from the center region of each split plot (11 m length) by avoiding 1 1 m along the outside length and 3 m along the width of each sampling area. Two cores (5 cm in diameter) were taken and divided in two depths (0–15 and 15–30 cm). The use of a larger core than the standard (2 cm in diameter) for soil testing was expected to reduce variability. In preliminary work (unpublished data), we have been able to achieve similar estimates of nutrient and labile fraction concentrations within plots by pooling two large cores (5 cm) as with six to eight small-sized 2.25 cm cores.

Samples from each depth were combined for use as a composite sample. In the laboratory, field moist samples were passed through a 25 mm sieve, then air-dried and ground to pass through a 2 mm sieve. A subsample was analyzed for P, K, Ca and Mg, total organic carbon (SOC), total nitrogen (TN) and pH by the Iowa State University Soil and Plant Analysis Laboratory. Phosphorus, K, Ca and Mg were determined for samples from the top 15 cm of depth as recommended on standard soil testing protocols. For P determination, Bray-1 P solution extractant was used[38]. Exchangeable K, Ca and Mg were extracted with an ammonium acetate solution. Soil pH was measured in a solution with a 1:1 soil to water ratio. SOC, TN and pH were determined for the 0–15 and 15–30 cm depths. SOC and N were evaluated by dry combustion analysis in a Costech ECS 4010 Elemental Combustion System.

Particulate organic matter C (POM-C), fluorescein diacetate (FDA) hydrolysis (a measure of soil heterotrophic activity), potentially mineralizable N (PMN) and hydrolysable amino $N + NH_4$ (IL-N) were determined in samples from 0 to 15 and 15 to 30 cm depths in the University of Illinois Soil Ecology Laboratory. For POM fractionation, we modified the method described in Marriott and Wander[7]. Briefly, 10 g air-dried soil was weighed into a 30 ml Nalgene plastic bottle. To the sample, 30 ml 10% (w/v) sodium hexametaphosphate solution was added and allowed to sit overnight to facilitate dispersion. The mouth of the 30 ml bottle was then covered with a 53-μm-mesh fabric and placed into a 250 ml Nalgene centrifuge bottle to which an additional 120 ml sodium hexametaphosphate solution was added. Samples were then shaken in a reciprocal shaker to allow separation of the mineral fraction from POM. After 1 h, the fine particles and sodium hexametaphosphate were discarded, and 150 ml tap water was added. Samples were shaken for 10 min to rinse the excess sodium hexametaphosphate and fine particles smaller than 53 μm. This rinsing procedure was repeated five times. The sample remaining in the 53 μm mesh fabric was considered POM. Using tap water, the POM was transferred to another piece of 53-μm-mesh fabric and oven dried at 50°C for 24 h. Dried samples were ground in a disk mill and analyzed for their C and N content (dry combustion analysis in a Costech ECS 4010 Elemental Combustion System).

Microbial enzymatic activity was evaluated with a modified version of the FDA hydrolysis method described by Schnurer and Rosswall[39] as improved by others[40,41] and optimized in our laboratory to achieve maximum

color development for our soils. Briefly, 1 g air-dried soil (ground <2 mm) was placed in a 50 ml Nalgene centrifuge tube. Twenty milliliters of sodium phosphate buffer (60 mmol l^{-1}, pH 7.6) were added, followed by 0.50 ml FDA stock solution (4.8 mmol l^{-1}). Bottles were tightly capped and shaken in an orbital shaker for 5 h at 200 rpm and 20°C. Then, 20 ml acetone was added and swirled to stop the reaction before the solution was vacuum-filtered through Whatman No. 4 filter paper. Controls were prepared for each soil type in the experiment following the same procedure as for FDA hydrolysis except that 0.50 ml acetone was added instead of 0.50 ml FDA stock solution. Blanks containing only the solutions were used to adjust the absorption curve for color development not associated with the soil samples. The concentration of FDA was estimated based on spectrophotometric readings of absorbance at 650 nm.

PMN was determined by anaerobic incubation with the method described by Drinkwater et al.[42] with small modifications. Two sets of 5 g samples of air-dried soil (ground <2 mm) were placed in 50 ml centrifuge tubes. Fifty milliliters 2 mol l^{-1} KCl were added to the first set to quantify the amount of NH_4^+ originally present in the soil. Samples were shaken for 1 h on a reciprocal shaker and vacuum-filtered through Whatman No. 42 filter paper. To the second set, 10 ml deionized water were added and incubated for 7 days at 40°C. After 1 week, samples were mixed with 40 ml 2.5 mol l^{-1} KCl solution, shaken, and the solution filtered in the same manner as the first set of samples. Ammonium concentrations in both sets were colorimetrically determined[43].

Hydrolysable amino N + NH4 was estimated using the Illinois-N (IL-N) procedure described by Khan et al.[44]. Air-dried soils (ground <2 mm) were treated with 2 mol l^{-1} NaOH in a tight glass jar and heated to 48 to 50°C for a 5-h period to liberate NH_4^+ and amino sugar-N as NH_3. Ammonia was collected in a 4% H_3BO_3 solution and its concentration was determined by semi-automated titration with 0.01 mol l^{-1} H_2SO_4 solution.

Statistical analysis and interpretation

Results from this experiment were analyzed in a split-split plot arrangement with farming systems as the main plot and the organic amendment and time in transition as the split plots using the PROC MIXED procedure[45]. A split–split–split plot design was used to evaluate the effect of depth (0–15 and 15–30 cm). The influence of soil variability was captured using four blocks that were treated as random effects. Variables that did not meet the assumptions of normality and homogeneity of variances were transformed prior to analysis. Potassium, Ca, Mg, pH, SOC, POM-N and POM C:N were log-transformed whereas TN, POM-C, FDA and PMN were square-root transformed. Two- and three-way interaction terms with P values >0.25 were dropped from the model and were not reported. Significant mean effects were separated with a least-squared means post hoc test adjusted with the Tukey adjustment to control type I error ($\alpha = 0.1$). Means of transformed variables were back-transformed for presentation. A CV was generated for each variable based on field replicates among experimental units. To evaluate the power of the analysis, we evaluated the minimum detectable difference as proposed by Yang et al.[46] and Stroup[47].

Indicators of biologically based fertility (SOC, TN, C:N ratio, POM-C, POM-N, POM C:N ratio, FDA, PMN and IL-N) were evaluated for their suitability for routine soil testing. We considered the time required for processing, costs for required laboratory equipment and the environmental/health risks associated with analyses. Evironmental/health risk assessment was based on the toxicity specifications for handling and disposal of chemical supplies[48].

To support indicator interpretation we adapted existing scoring functions for pH, available P, SOC and PMN from Andrews et al.[29] which are described by the following equations:

For pH:

$$y = \frac{a \times \exp[-(pH - b)^2}{2 \times c^2} \quad (1)$$

where $a = 1.0$ and $b, c = \int(\text{crop})$

For available P:

If $P \leqslant$ max (for crop and method) $\Rightarrow$

$$y = \frac{(a \times b + c \times P^d)}{b + P^d} \quad (2)$$

If $P >$ max (for slope and method) $\Rightarrow$

$$y = a - b\exp(-c \times P^d) \quad (3)$$

Else $y = 1$, where $a = 9.26 = 10^6$; $c = 1.0$; $d = 3.06$ and $b = \int(\text{crop, SOC, texture, method})$

For SOC:

$$y = \frac{a}{1 + b \times \exp(-c \times \text{SOC})} \quad (4)$$

where y is the indicator score, $a = 1; b = 50.1$ and $c = \int(\text{iOM, texture, climate})$

For PMN:

$$y = \frac{a}{1 + b \times \exp(-c \times \text{PMN})} \quad (5)$$

where $a = 1$; $b = 50.1$ and $c = \int(\text{iOM, texture, climate})$

We set agronomic thresholds using recommendations for horticultural and agronomic crops grown within our region[49]. Environmental thresholds were based on the scoring functions suggested by Andrews et al.[29], where the indicator concentration was transformed to a unitless scale from 0 to 1. A value closer to zero indicates that the system is functioning at lowest capacity, whereas values

Table 2. The influence of management system, organic amendment, depth of sampling and time under transition on soil attributes used for soil testing and soil organic matter. In the table are *P* values for main effects and interaction terms in an analysis of variance (ANOVA).

Factor/dependent variable[1]	P	K	Ca	Mg	pH	SOC	TN	C:N ratio
P values								
System	0.55	0.58	0.55	0.59	0.44	0.63	0.55	0.99
Amendment	0.80	0.98	0.36	0.77	0.10	0.78	0.52	0.66
Transition	0.04	<0.001	<0.001	<0.001	0.69	0.08	<0.001	0.001
Depth	ND[3]	ND	ND	ND	<0.001	<0.001	<0.001	<0.001
System × amendment	D[2]	D	D	D	0.1512	D	D	D
System × transition	D	D	D	D	D	D	0.18	D
Amendment × transition	D	D	D	D	D	D	0.06	D
System × depth	ND	ND	ND	ND	D	D	D	D
Amendment × depth	ND	ND	ND	ND	D	D	0.23	D
Transition × depth	ND	ND	ND	ND	0.09	0.26	D	0.02
System × amendment × transition	D	D	D	D	D	D	D	D

[1] P, = available P; K, exchangeable K; Ca, calcium; Mg, Magnesium; SOC, total organic C and TN, total N.
[2] Interaction terms with *P* value >0.25 were dropped (D) from the model.
[3] P, K, Ca and Mg were only determined in the 0–15 cm depth.

Table 3. Changes in standard test values of available P, exchangeable K, Ca, Mg and pH during the three-year transition period toward organic certification. Values represent means across all systems and amendment types ($n=4$). Samples were collected at the Windsor Organic Research Trial in the spring of 2003 and 2006.

Soil property	Depth (cm)[1]	2003	2006	% CV
Available P (Bray-P) ($mg\,kg^{-1}$ soil)	0–15	53b[2]	62a	30
Exchangeable K ($mg\,kg^{-1}$ soil)	0–15	162b	247a	40
Ca ($mg\,kg^{-1}$ soil)	0–15	2186b	3041a	23
Mg ($mg\,kg^{-1}$ soil)	0–15	233b	311a	33
pH[3]	0–15	6.76	6.81	55[4]
	15–30	6.64	6.55	

[1] Samples were analyzed at the 0–15 cm depth for P, K, Ca and Mg. In the case of pH it was quantified at the 0–15 and 15–30 cm of depth.
[2] Means followed by different letters in the same row are significantly different at $\alpha=0.1$.
[3] Time by depth (transition × depth) interaction was not significantly different at $\alpha=0.1$.
[4] Coefficient of variation for the concentration of hydrogen ions [H^+].

closer to one approach the system's maximum capacity. In addition, we compiled reports from the scientific literature for soils managed using organic and, when necessary, conventional cropping practices. Our literature survey was limited to studies conducted on silt and clay loam soils in regions with temperate climates.

Results and Discussion

Management effects on soil test values and indicators of biologically based fertility

Our prediction that soil quality would be most improved where disturbance was low and organic additions were high was not met. Furthermore, no significant differences were detected among systems, but significant increases were detected in the concentrations of all nutrients and labile organic matter fractions after the 3-year transition (Tables 2 and 3). Even though the vegetable-based system was tilled 6 times more frequently than the ley-based system and 1.5 times more frequently than the row crop system, there was no evidence of significant difference in soil quality among systems. This may have been due to the lack of power in our statistical design to detect significance. Cumulative compost and manure additions (which on average contained 545, 448 and 364 kg of N, P and K ha^{-1}, and 476, 163 and 289 kg N, P and K ha^{-1}, respectively) did not significantly increase soil reserves over those found in the untreated plots, which relied on N derived from N fixation and on P and K recycling from residues or captured/remobilized by roots. Based on estimates of nutrients removed in harvested crops, which were ranked row-crop > vegetable > ley, reductions in K (and to a lesser extent P) concentrations in unfertilized plots had been expected. Instead, the average concentrations of available P, K, Ca and Mg in baseline (pre-transition) samples were 51, 162, 2186 and 233 $mg\,kg^{-1}$ soil, respectively. After transition, nutrient concentrations in the 0–15 cm depth had increased to 62, 247, 3041 and 311 $mg\,kg^{-1}$ soil, respectively. Soil pH (0–15 cm) did not

Table 4. The influence of management system, organic amendment, depth of sampling, and time under transition on attributes of labile soil organic matter. In the table are *P* values for main effects and interaction terms in an analysis of variance (ANOVA).

Factor/dependent variable[1]	POM-C	POM-N	POM C:N ratio	FDA	IL-N	PMN
P values						
System	0.34	0.90	0.36	0.90	0.81	0.61
Amendment	0.63	0.62	0.04	0.64	0.91	0.80
Transition	0.03	<0.001	0.10	<0.001	0.003	<0.001
Depth	<0.001	<0.001	0.20	<0.001	<0.001	<0.001
System × amendment	D[2]	D	D	D	D	D
System × transition	D	0.24	D	D	D	D
Amendment × transition	D	0.12	D	D	D	D
System × depth	D	D	D	D	D	0.23
Amendment × depth	D	0.17	0.15	0.16	D	D
Transition × depth	0.24	D	0.22	0.04	D	0.04
System × amendment × transition	D	D	D	0.12	D	D

[1] POM-C, particulate organic matter C; POM-N, particulate organic matter N; FDA, fluorescein diacetate hydrolysis; IL-N, Illinois N test and PMN, potentially mineralizable N.
[2] Interaction terms with *P* value >0.25 were dropped (D) from the model.

change significantly during transition but trended upward even though no lime was applied (Tables 2 and 3). All test values excluding pH, were in the 'very high' soil test range on both sampling dates[50]. Concentrations were similar to those reported by Delate and Cambardella[51] in a four-year-old organic row crop system that was also conducted on an inherently productive prairie soil. Their work, and that of Clark et al.[52], which was conducted on vegetable-based systems, reported increased nutrient contents in soils after organic transition in systems amended with compost or manure and reductions when amendments were not applied. Declining test values have also been reported in several European studies when systems did not include application of manure or compost[5,34,53]. Smukler et al.[54] reported slight reductions in P and K test values during a 3-year transition period in systems amended annually with compost and manure. That study considered a more intensive rotation that allowed for at least two crop harvests per growing season. We attribute our results to intensive nutrient recycling and active rooting by cover crops that increased nutrient availability and recovered nutrients from the subsoil to enrich the surface. This kind of increase in availability in the rhizosphere or where organic amendments are applied has been widely observed[55,56]. Root-based nutrient concentration has been observed by others, including Thorup-Kristensen et al.[57,58] who found that by adding cover crops to a wheat system they were able to capture and recover enough nutrients to offset losses from crop removal. Several reports of root-based concentration of nutrients within the soil surface appear in studies conducted on inherently fertile soils[59,60].

Although SOC increased slightly over time ($P<0.1$), increases were not significantly different when the interaction effects with system, amendment regime or depth were taken into consideration (Table 2). Despite the lack of statistical significance, the overall increase (see Table 5) is consistent with other works showing gains in organic matter under organic management[7,51,52,54,61]. Increased concentration in TN was most pronounced in the deeper soil depth and contributed to a reduced post-transition soil C:N ratio (Tables 2 and 5). This change in soil C:N ratio suggests N addition to the subsurface, rather than solely organic matter decay, and may reflect inputs from leguminous species with more readily decomposable substrates[7].

The similarity among soils transitioned under different organic production systems was surprising given that the more dynamic properties are known to be quite responsive to manure and compost additions. The statistical separation between pre- and post-transition soils was greater for labile soil organic matter fractions than for total SOC (Tables 4 and 5). This illustrates why these assays are used as indicators of change and is consistent with other works showing that statistically significant differences will emerge more rapidly within labile soil organic matter fractions than in whole soil C or N stocks[3,62,63]. Sensitivity of indicators will be discussed later. The magnitude of increase in POM concentrations observed was comparable to those seen in other US organic systems[7]. In a comparison of organic and conventional systems, Marriot and Wander[7] found that POM-C contents were similar in manure- and legume-based systems and low in conventional systems. We found POM C:N ratios varied only slightly after transition ($P=0.1$), and were similar in the compost amended and cover cropped plots (22) but greater than POM C:N ratios observed in plots amended with manure (20). In general, POM C:N ratios exceeding 20 reflect inputs from carbonaceous plant residues and indicate use of soil-building practices; ratios approaching 15 are commonly associated with an abundance of decayed organic matter but can briefly occur after incorporation of fresh legume-derived residues[3,63]. To evaluate the relationship between management practices and POM one must consider both the quantity and quality of the fraction.

Table 5. Changes in SOC, TN, C:N ratio and measures of biologically based fertility (POM-C, POM-N, POMC:N ratio, FDA, IL-N and PMN) as affected by time of organic management and sampling depth ($n=4$). Samples were collected at the Windsor Organic Research Trial at two sampling depths.

Soil property	Depth (cm)[1]	2003	2006	% CV	% change
Mean estimates					
SOC ($g C kg^{-1}$ soil)[3]	0–15	21.9	23.4	20	7
	15–30	18.9	19.6	22	4
TN ($g N kg^{-1}$ soil)[3]	0–15	1.7	1.8	16	6
	15–30	1.5	1.6	19	9
C:N ratio	0–15	12.7a[2]	12.7a	7	(<0.1)[4]
	15–30	12.2a	11.8b	12	(3)
POM-C ($g C kg^{-1}$ soil)[3]	0–15	2.9	3.1	24	6
	15–30	1.2	1.4	37	20
POM-N ($g N kg^{-1}$ soil)[3]	0–15	0.1	0.1	26	8
	15–30	0.05	0.06	33	20
POM C:N ratio[2]	0–15	21.9	22.2	11	1
	15–30	20.7	22.1	19	7
FDA ($ug FDA g^{-1} soil h^{-1}$)	0–15	29.6b	51.6a	47	74
	15–30	14.4d	25.0c	59	74
IL-N (mg NH_4-N kg^{-1} soil)[3]	0–15	268.1	259.6	13	(3)
	15–30	253.2	238.9	17	(6)
PMN (mg NH_4-N kg^{-1} soil $week^{-1}$)	0–15	34.8a	40.2a	47	15
	15–30	20.1b	33.0a	51	64

[1] Samples were collected at the 0–15 and 15–30 cm of soil profile.
[2] Means followed by different lower case letters within the same variable are significantly different at $\alpha=0.1$.
[3] Time × depth (transition × depth) interaction was not significantly different at $\alpha=0.1$
[4] Numbers in parentheses mean a negative change over time.

Increases in FDA hydrolysis rates and PMN concentrations paralleled those of POM and were positively related. The increase in FDA activity after transition (74%) was more pronounced than for POM or PMN (13 and 40%, respectively) (Table 5). Drinkwater et al.[4] also found FDA to be positively related to measures of available N and argued that this showed how organic practices increase N supply by promoting microbial activity. Unlike PMN, the IL-N, which is a biochemically based method used to estimate mineralizable N, declined during transition. In their multi-site study of organic systems, Marriott and Wander[7] found that the IL-N was not as sensitive to different organic management practices as POM-N and was more closely related to clay content than these other estimates of mineralizable N. That result agrees with the findings of Spargo et al.[64] who found that the IL-N was closely related to TN. In this work, IL-N declined with time, whereas TN increased, suggesting that the proportion of N in the hydrolysable fraction was reduced during transition but converted to another, possibly more resistant form. According to Kwon et al.[65], the IL-N test detects N in only some components of the microbial biomass. They showed that the IL-N recovered 95% of monomeric amino sugars contained in bacterial cell walls and 55% of amides abundant in fungi and concluded that the assay did not recover alpha amino acids or chitin. Further, they showed that the IL-N test was more sensitive to changes in bacterial amino sugars than to fungal amides and noted that this would complicate interpretation in systems that promote fungal biomass. This might suggest that the proportion of N in amides increased with transition and provided indirect evidence of an increased importance of fungal biomass in these post-transition soils.

Significant depth-by-date interactions revealed important changes in soil quality (Tables 4 and 5). As expected, all variables were more concentrated in the surface than subsurface depths; however, the degree of stratification for most indicators of biologically based fertility (SOC, POM-C, POM-N and PMN) decreased during transition (Table 5). Stratification by depth was less pronounced for pH than for other variables (Table 3). The depth distribution of soil organic matter in the ley system was similar to that found in the tilled row crop- and vegetable-based systems even though it was not tilled after seeding; that occurred within a month of our baseline sample collection. Other works have shown that use of perennial mixtures or cover crops and manure increase the depth of soil organic matter enrichment[66,67]. We attribute study-wide reductions in soil organic matter stratification to active C additions principally associated with diversification of the crop rotation. According to Kong and Six[68], it is likely that the presence of active roots in the systems is more important for soil organic matter accumulation than the incorporation of above-ground residue and/or organic amendments. These results suggest organic farmers should sample more deeply than the current

Table 6. Estimates of labor and cost of laboratory equipment needed to process soil samples for SOC, TN, C:N ratio, POM-C, POM-N, POM C:N ratio, FDA, IL-N and PMN.

Variables	Labor (h/sample)	Equipment ($ approximate cost)	Materials
SOC + TN + C:N ratio	0.22	Analytical balance ($ 4500), grinder ($10,000), CHN analyzer ($ 26,000)	Acetanilide standard (1)[5], chromium oxide (2), cobaltic oxide (3), magnesium perchlorate (1), cupper wire (1), helium (2)
POM-C + POM-N + POM C:N ratio	0.32[1]	Analytical balance ($ 4500), reciprocal shaker ($ 2200), grinder ($ 10,000), CHN analyzer ($ 26,200)	Sodium metaphosphate (2), acetanilide standard (1), chromium oxide (2), cobaltic oxide (3), magnesium perchlorate (1), cupper wire (1), helium (2)
FDA	0.11[2]	Analytical balance ($ 4500),orbital shaker ($ 2200), microplate reader ($ 5600)	Sodium phosphate tribasic (2), acetone (2) and FDA (1)
IL-N	0.25[3]	Analytical balance ($ 4500), hot plate apparatus ($ 900), titrator ($ 5300)	Sodium hydroxide (4), boric acid (2), sulfuric acid (4)
PMN	0.12[4]	Analytical balance ($ 4500),reciprocal shaker ($ 2200), microplate reader ($ 5600)	Potassium chloride (2), 2-hydroxibiphenyl sodium (3), sodium nitroprusside (3), trisodium citrate (1)

[1] Labor estimates assuming samples were processed in sets of 30 for POM fractionation and sets of 35 for CN analysis.
[2] Labor estimates assuming samples were processed in sets of 80.
[3] Labor estimates assuming samples were processed in sets of 33.
[4] Labor estimates assuming samples were processed in sets of 50.
[5] Numbers in parenthesis refer to the health risk hazard from the material safety data sheets for each compound. Being 1, slight; 2, moderate; 3, severe and 4, extreme.

norm (0–15 cm), at least in Mollisols in Illinois, to allow them to better manage the active rooting zone.

Sensitivity of soil organic matter attributes to changes in soil resource condition

The relative utility of the indicators of biologically based fertility depends on each measure's responsiveness to management, inherent variability, costs and environmental risks associated with each method. We assessed indicator sensitivity by considering the relative change in soil organic matter fractions with respect to baseline samples (control samples collected before any organic management was established). Soil FDA, PMN, POM-N and POM-C contents were the variables with the greatest percent change observed during transition (Table 5). The consistency of each soil organic matter fraction was assessed using its variability and represented by the coefficient of variation (CV, %). Coefficients of variation for FDA, PMN and POM fractions observed in both depths ranked FDA > PMN > POM-C and POM-N. The lowest CVs were found for SOC, TN, IL-N and the POM and soil C:N ratios (SOC > TN > IL-N > POM C:N ratio > soil C:N ratio). Even though FDA, and hence heterotrophic activity, was quite responsive to management, this parameter was also highly variable and this decreased the power of the test to around 20% for the farming system by transition interaction. Based on FDA's variability we would have needed to collect ten composite samples per management system in order to detect treatment differences with a 90% confidence level. Clearly, this is impractical given the limited amount of resources devoted to establishment of replicated field experiments, sampling and testing. The POM fraction and SOC were less variable than FDA and so would have required about six and eight replicate samples to be taken per treatment to document differences at the 90% confidence level. The variability of soil organic matter fractions in both depths was similar to that observed for chemical test variables in the 0–15 cm depth. This suggests that testing regimes that incorporated indicators of biologically based fertility would not need to modify sampling intensity even if they were altered to increase sampling depth. Further work is needed to explore indicator variability in other soils and organic cropping systems.

The potential for commercial use of biologically based indicators was assessed using the costs associated with labor and equipment and the health-risk-hazard of chemical use. All measurements considered in this study were affordable (< 0.5 h of labor per sample) and so deemed suitable for use in commercial soil testing laboratories. Determination of indicators like FDA, IL-N and PMN does not require specialized equipment. Labor requirement was 0.1–0.3 h per sample (Table 6) assuming that sets of multiple samples could be processed simultaneously. Soil FDA hydrolysis rate was the least—and POM was the most—labor-intensive methods described here. We note that methodological improvements could reduce time required to recover the POM fraction. These labor requirements are similar to those needed for determination of conventional chemical soil tests (G. Steffen, Illinois Soil Testing Association, pers. comm., 2008). The chemical supplies used during processing

Table 7. Reference values for indicators of soil organic matter found in the organic literature.

Soil property	Management	(No. of sites)	Median	Concentration (Min–Max)	Citations
g kg^{-1} SOC (g C kg^{-1} soil)	Organic	13	16.7	10.0–28.0	Drinkwater et al.[4]; Marriott and Wander[7]; Khan et al.[44]; Delate and Cambardella[51]; Wander et al.[61]; Teasdale et al.[66]; Wander et al.[67]; Wander and Bollero[83]; Needelman et al.[84]; Poudel et al.[85]; Maughan et al.[86]; Varvel and Wilhelm[87].
	Conventional	14	16.2	9.0–26.0	
TN (g N kg^{-1} soil)	Organic	9	1.5	0.9–2.4	Drinkwater et al.[4]; Marriott and Wander[7]; Khan et al.[44]; Delate and Cambardella,[51]; Wander et al.[61]; Teasdale et al.[66]; Wander et al.[67]; Wander and Bollero[83]; Needelman et al.[84]; Poudel et al.[85]; Maughan et al.[86]
	Conventional	13	1.2	0.9–1.9	
POM-C (g C kg^{-1} soil)	Organic	7	2.9	2.0–5.8	Marriott and Wander[7]; Wander et al.[67]; Maughan et al.[86]; Mirsky et al.[88]; Coulter et al.[89]; Hernandez-Ramirez et al.[90]
	Conventional	7	2.2	1.8–2.5	
POM-N (g N kg^{-1} soil)	Organic	2	0.2	0.2–0.2	Marriott and Wander[7]; Maughan et al.[86]; Coulter et al.[89]; Hernandez-Ramirez et al.[90]
	Conventional	5	0.1	0.1–0.2	
POM C:N ratio	Organic	2	19.0	15.0–23.0	Marriott and Wander[7]; Maughan et al.[86]; Coulter et al.[89]; Hernandez-Ramirez et al.[90]
	Conventional	5	18.0	15.0–18.0	
FDA (μg FDA g^{-1} dry soil h^{-1})	Organic	4	23.0	18.0–66.0	Drinkwater et al.[4]; Workneh and van Bruggen[91]; van Bruggen[92]; Mendes et al.[93]; Darby et al.[94]
	Conventional	5	20.0	12.0–149.0	
IL-N (mg NH_4-N kg^{-1} soil)	Organic	2	334.0	165.0–503.0	Marriott and Wander[7]; Khan et al.[44]; Spargo et al.[64]; Wander et al.[67]; Ruffo et al.[95]
	Conventional	18	224.0	98.0–465.0	
PMN (mg NH_4-N kg^{-1} soil week^{-1})	Organic	8	6.4	4.5–29.4	Drinkwater et al.[4]; Needelman et al.[84]; Clark et al.[96]; Liebig and Doran[97]; Idowu et al.[98]
	Conventional	10	4.7	4.3–22.0	

would fall into the moderate health rating hazard[49]. Concentrations of those compounds that rank under the high health risk hazard were used in very small quantities and very low concentrations; thus, safety would not be a concern with the use of normal laboratory precautions for protection and handling and with proper disposal.

Interpretation of soil testing values

To evaluate the influence of organic transition on soil quality at our site we used conventional testing and then sought to adapt interpretation guides developed for conventional agriculture and included appropriate scoring functions. Currently, there are no critical values or proposed methodologies of interpretation for most indicators of biologically based fertility with exceptions for SOC[29], PMN[29] and IL-N[44].

Table 7 includes a summary of the median, maximum, and minimum values for variables found in the relevant literature. These values describe the general range of concentrations one could expect to find in medium- to fine-textured soils under organic and conventional management practices. Data from organic systems represent a wider range of tillage and fertilization practices applied in cash grain and vegetable production systems; this probably accounts for the wider range of values found for SOC, TN, POM and PMN fractions. The small number of reports for FDA and IL-N in organic systems explains the relatively narrower range. In general, concentrations of indicators used in our study approached the upper limit for reported values. As previously noted, standard soil test values were also in the high-to-very-high range. This suggests that nutrients and probably energy reserves were abundant or in surplus at this site. Concentrations of PMN (40 mg NH_4–N kg^{-1} soil week^{-1}) were significantly higher than those reported in the literature and our mean IL-N was above the value (235 mg NH_4–N kg^{-1}) Khan et al.[44] found to be a critical limit beyond which corn would not respond to additional N fertilization. The IL-N critical limit should be relevant to other N-demanding crops such as tomato or broccoli[49].

Figure 1 presents some of our data using the soil scoring curves proposed by Andrews et al.[29]. Both pre- and post-transition observations fell on the asymptote of the scoring functions for pH, available P and PMN and on the ascending portion of the SOC curve. Based on standard interpretations, the high scores suggest that these soils were functioning near their maximum capacity for productivity. It may be appropriate, however, to reduce the level of the plateau used in the functions when applied within organic systems or to apply an optimum-shaped function, which declines past some maximum because the presence of high levels of available nutrients could suppress plant–microbe associations contributing

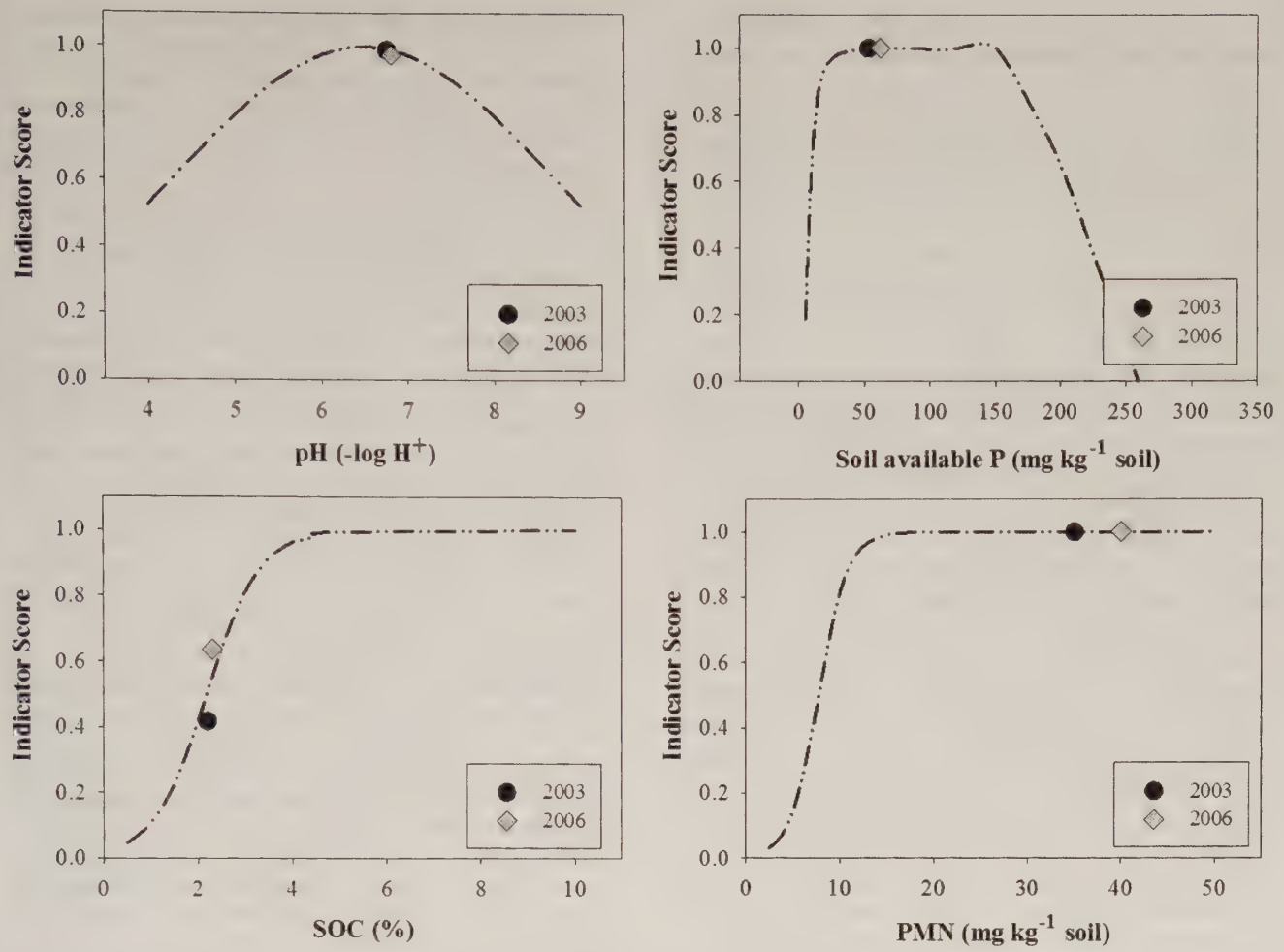

Figure 1. Scoring functions illustrating the change in scores affected by organic transition. Dotted lines represent the expected values for soils classified as Catlin silt loam and Drummer silty clay loam in Champaign, IL using the algorithms presented in Andrews et al.[29]. Symbols within the curve represent the mean average scores for each indicator computed in our study on soils collected pre- and post-transition. Black circles are the means for samples collected in 2003 before organic transition started; gray diamonds are the means for samples collected in 2006 after organic transition ended. Indicators were: pH, available P, SOC and PMN at the Windsor Organic Research Trial. Scores for PMN and SOC assume a more-is-better relationship, with no reduction for environmental concerns after they reach a plateau. Soil pH is an optimum function that depends on the crop in place.

to tight nutrient cycling[69,70]. The shape of the P curve already takes environmental excess into account and reduces scores when threats to water quality emerge. It is not clear whether this is an appropriate model for available N, which in our study is in the high-to-very-high range for both PMN and IL-N compared to other published works. Organic matter stocks were low compared to the site's potential as a result of the history of cultivation[71]. Values on the ascending section of the SOC curve suggest that further increases in SOC would enhance soil function by enhancing soil physical properties and nutrient availability but further increases in the concentration of labile N, estimated by PMN or IL-N would not be desirable from an environmental perspective. There is not a simple linear relationship between C inputs and SOC accumulation[72], and soil N status can alter decay dynamics[67,73]. Ideally our soils could continue to accumulate organic C and with it, N in forms less prone to undergo nitrification and subsequent loss, but these forms would be difficult to separate and quantify[74]. The small increase observed in the concentrations of POM-C, which was from 7.5 to 13% of SOC, could be due to positive priming of C caused by high levels of available N[75]. Limitation of SOC accrual has been observed in systems where liberal or chronic applications of N are made in the form of manure[67,76] or inorganic fertilizers (e.g., Shen and Bartha[77], Gerzabek et al.[78], Bol et al.[79], Van Lauwe et al.[80] and Blagodatsky et al.[81] as reviewed by Kuzyakov et al.[75] and Khan et al.[82]). This is why we propose changing the plateau function to an optimal scoring function for PMN with reductions in scores occurring when levels exceed an optimum. To account for the microbial contribution to N availability, the scoring function should consider both C and N status. If we are correct in our theory that reductions in IL-N accompanied by increases in PMN signal a shift in microbial N

explained by increased fungal importance, then we have moved left along the curve toward a more optimal condition. This ability to decline or maintain available N even while total N increases might indicate greater potential for the IL-N or other amino-sugar-based assays to be used as indicators. It is doubtful that PMN would fall unless SOC and TN levels fell too.

Conclusions

In general, organic management increased standard and biological soil test values observed in soils under organic perennial ley-, row crop- and vegetable-based production. Regardless of management intensity, organic practices improved/maintained soil resource condition in the plow layer. Increases in resource abundance in the 15–30 cm depth were notable and suggest that deeper sampling might be needed to better track changes in the soil resource caused by organic practices.

Data were used to demonstrate how indicators of biologically based fertility could have practical and therefore commercial application, and outline a strategy to use scoring functions to create an interpretive framework for assessment of performance in organic cropping systems. By revising established norms and values published in the literature, we concluded that soil nutrient stocks were at or beyond optimum agronomic levels and may be too high from an environmental perspective. Intermediate scores for SOC and modest rates of SOC and POM-C accrual suggest that reductions in labile N may be needed. The results suggest that scoring functions for biologically based indicators of soil quality should be optimum functions and seek to account for feedbacks influencing biotic interactions.

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Sulfur management and miticide use in winegrapes grown in California

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Abstract

Proper sulfur management in winegrapes may potentially reduce both insecticide/miticide use and the human/social problems associated with sulfur's impacts on air quality and odors, particularly in wine tourism regions. Data from California's unique Pesticide Use Report (PUR) database, which records agricultural pesticide applications on all crops throughout the state, are used to determine if either winegrape grower's choice of sulfur formulation (dust only, wettable only, or a combination) or overall sulfur use rates (pounds per acre planted) correlate with annual miticide and insecticide use in Fresno and Madera (hotter–drier) and Napa and Sonoma counties (cooler–damper). Annual sulfur use has declined by 36–55% in these counties from 1993 to 2009. In 2000, the greatest number of growers in each county were combination users; wettable users were higher in Napa (38%) and Sonoma (34%) than in Fresno (24%) and Madera (10%); and dust-only users varied little (15–19%) across the four counties. Data for 2005 and 2009 showed similar trends. The use of high-toxicity insecticides in Fresno was 387% higher than in Napa–Sonoma in 2000, but was 25 and 8% lower in 2005 and 2009, respectively. In Fresno, wettable sulfur users used less high-toxicity insecticides; while in Fresno and Madera dust users used less lower-risk insecticides than combination or wettable sulfur users. No significant differences in insecticide use were evident between the three sulfur use categories in Napa–Sonoma ($P=0.97$). On average, dust users in Fresno–Madera used more high-toxicity miticides than combination or wettable sulfur users in 2000. That trend decreased in the data for 2005 and 2009. Average miticide use in Fresno–Madera was higher than in Napa–Sonoma by 1349% in 2000, 1103% in 2005 and 146% in 2009. Higher sulfur use intensities among individual growers in Fresno–Madera were positively correlated with greater use of high-toxicity miticides in 2000 and 2005, but not 2009. The comparable results for Napa–Sonoma were less clear. Since PUR data represent statewide information on pesticide use decisions in real-world farming scenarios, it can complement data from field and laboratory studies. Expanding these analyses to compare pesticide use by individual growers across years, or to correlate pesticide usage with any published mite field surveys may shed more light on the enigmatic relationship between sulfur fungicide use and mite outbreaks in winegrapes.

Key words: sulfur management, miticide use, winegrapes

Introduction

Winegrapes are a significant crop in California. They are grown in 47 of California's 58 counties, cover 477,000 acres among 4800 winegrape growers, and rank among the State's top ten agricultural products[1]. California is one of the world's leading grape-producing regions, and its winegrape community had a statewide economic impact of $2.76 billion in 2004[2], accounting for 90% of US grape production. The success of the industry, however, depends on the use of pest control strategies to manage the insects, mites and diseases that threaten the crop every year. In order to maximize crop production, California growers use many pounds of sulfur and other fungicides to control disease and apply a variety of miticides and insecticides to control mites and insects, respectively.

Sulfur is the leading pesticide active ingredient used on winegrapes in California, accounting for 95–98% of the total weight of pesticide used on the crop[3]. Although sulfur is very effective in controlling grape powdery mildew, *Uncinula necator* (Schw.) Burr., and is a relatively safe[4] natural element in terms of its environmental toxicity, the use of sulfur can degrade air quality, create foul odor, cause irritation of the eyes and skin, and result in breathing difficulty. A survey by the California Department of Pesticide Regulation (CDPR) found

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86 reported incidents[5] of sulfur drift from 1997 to June 1999. Two-thirds of the reports cited grapes as the target source and 80% were attributed to sulfur dust applications. Similarly, Browde and Ohmart[6] reported that a high percentage of drift incidents were attributable to sulfur dust applications. The susceptibility of sulfur dust to offsite movement by wind creates conflicts with the human and social environment, particularly as residential areas move closer to agricultural lands, increasing the agriculture/urban interface. Despite the drawbacks of sulfur use, growers rely on sulfur for disease control because of its low cost, ease of application and proven lack of resistance of the major fungal pest of grapes: powdery mildew.

Many studies have linked frequent applications or high use rates of sulfur dust to a reduction in predatory mite populations in grape-growing regions around the world, including California and the Pacific Northwest. In a single-season study of a vineyard near Madera, Hanna et al.[7] found that regular sulfur applications for powdery mildew control can exacerbate spider mite problems in vineyards by suppressing populations of predatory thrips and predatory mites. Research by Calvert and Huffaker[8] found a consistent correlation between overwintering field populations of the spider mite predator, Western predatory mite *Galendromus* (= *Metaseiulus*) *occidentalis* and number of sulfur treatments. Lower predatory mite populations were found in the ten-treatment sulfur plots than in the random, control, pollen treated and four-treatment sulfur plots. In a study of two Thompson seedless raisin-producing vineyards in southern Madera county, English-Loeb et al.[9] found that Pacific and Willamette mite pests became abundant on vines after sulfur treatment stopped, indicating that some form of sulfur disruption was occurring. Similar results have been obtained for predatory mites in other crops, such as hops in the Pacific Northwest[8]. On the other hand, some studies find no effect of sulfur on predatory mites[10].

Research in this field is now exploring possible mechanisms of mite outbreak stimulation by sulfur apart from its effects on natural enemy mortality. Costello and Albers[11] proposed that the observed sulfur disruption of mite populations may be due to a plant-based factor, with sulfur acting to suppress the ability of the plant to defend itself against mite attack, rather than by suppression of predatory mite populations[12]. Sulfur[13] and other fungicides[14] have been shown to kill another natural enemy of mites—acaropathogenic fungi. Another study found that sulfur use decreases not only populations of beneficial predatory mites but also their feeding behavior[15]. Grape powdery mildew, which is suppressed by sulfur treatment, may also be an important food source for some generalist predatory mite species which are important in vineyards[16]. A recent study found a greater effect of sulfur on decreasing fecundity than on increasing mortality in Oregon's dominant vineyard predatory mite, *Typhlodromus pyri*[17].

Other research has identified a broader relationship between the use of insecticides and disruption of natural mite controls. UC IPM[18] guidelines report that 'spider mites have been serious pests for several decades, and for reasons not yet fully documented, it is believed that use of synthetic organic insecticides has upset natural controls in some situations.' For example, Prischmann et al.[19,20] found that either chlorpyrifos, sulfur or combinations of the two dramatically reduced populations of several predatory mite species in Washington State vineyards. While toxic effects of many insecticides toward predatory mites are short-lived[21] some show prolonged persistence. For example, Zalom et al.[22] found that residues of permethrin and esfenvalerate were toxic to *G. occidentalis* for up to 7 months after application.

Improved sulfur management, with regard to amount, frequency and formulation of sulfur used, can not only reduce the incidence of drift and associated human exposures but may also allow for reduction of subsequent insecticide and miticide use to deal with sulfur-induced pest outbreaks. Sulfur management practices that maintain natural enemies of pest mites may be important parts of a strategy to reduce the use of the most highly toxic miticides, specifically those targeted before 2006 for reduction under the Food Quality Protection Act (FQPA).

This study uses a novel approach to examine the sulfur use patterns in the selected four counties and to shed some light on the relationship between sulfur use and toxicity of pesticides. Data from the Pesticide Use Report (PUR) database of the CDPR[23] are examined to identify whether sulfur product choice or application rates show a relationship to the total annual amount of applied miticides and insecticides in the major winegrape-growing regions of California. Sulfur management in coastal and valley counties of California was investigated to explore the relationship between sulfur and miticide use in these distinct winegrowing areas. The analysis of the relationship between sulfur and miticide/insecticide use focuses on specific active ingredients on the FQPA pesticide Priority Lists I & II[24] (hereafter 'FQPA I & II'), which indicates that they are targeted for reduction due to their high toxicity.

Materials and Methods

Study areas

Due to the wide climatic range of winegrape-growing regions in California, this study selected four counties (Fig. 1) to represent the distinctive valley and coastal weather conditions. Fresno and Madera were selected to represent the hotter and dryer inland valley wine-growing region, while Napa and Sonoma were selected to represent the more moderate and humid weather conditions of a coastal wine-growing region. It is important to note that microclimates within each region also

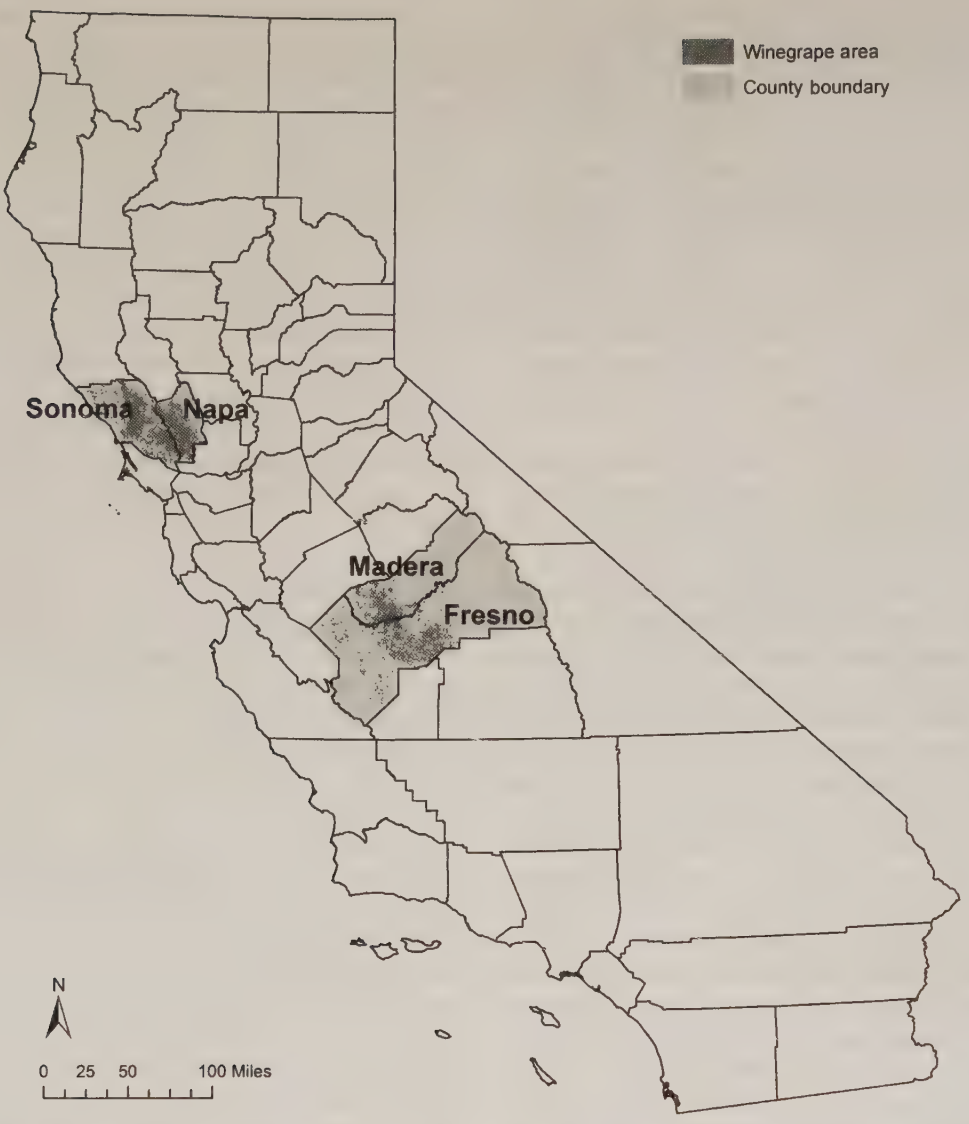

Figure 1. Study site, the four selected counties (Napa, Sonoma, Madera and Fresno) growing winegrape in the state of California.

have important effects on pest and disease pressures, and these local variations would affect pest management strategies.

Fresno county, located in the southern San Joaquin Valley, is the largest agricultural county in California, producing approximately 15% of all winegrapes grown in the State. Most of Fresno's agricultural lands are located in the western portion of the county. Fresno has hot and dry summers with average maximum temperature of ~96–98°F in July–August. Average annual rainfall is ~10 inches.

Madera county, situated in the heart of California's central valley, is another major agricultural county. It is bordered to the south by Fresno county and to the north by Merced and Mariposa counties. Winegrapes account for more than 45% of the agricultural acreage in Madera county and this county hosts 10% of the State's winegrape acreage. Similar to Fresno, Madera has warm and dry summers with an average maximum temperature in July of 99°F, though the hottest day in July of 2005 was over 110°F. Average annual rainfall is ~12 inches.

Napa county is in the heart of Napa Valley, 63 miles southwest of Sacramento and 47 miles northeast of San Francisco. This county is filled with world-renowned vineyards and surrounded by a rim of rolling hills and mountains of spectacular beauty. In 2006, the North Coast region, which includes both Napa and Sonoma counties, produced US$779 million of winegrapes[25]. Winegrape acreage in Napa county accounts for approximately 7% of California's total. The climate in Napa is mild, with an average maximum temperature in the summer of ~70°F, an average annual rainfall of 23.88 inches, and monthly humidity averages ranging from ~65 to 80% throughout the year.

Sonoma county is located on California's Pacific coast north of the San Francisco Bay Area. The value of Sonoma county winegrapes is approximately US$310 million annually, and its winegrape acreage accounts for

~12% of the State's total. The climate of the county is characterized by moderate temperatures and precipitation. Along the coast, temperatures remain cool throughout the summer, due to the great moderating influence of the ocean. From late summer to fall the average sea-water temperature is approximately 55°F. Throughout the county, average seasonal precipitation ranges from less than 20 inches in the extreme southeast corner to 30–40 inches over much of the central part to 70–80 inches in the mountain areas.

Based on the PUR data, in 2009 there were 800 growers growing more than 55,000 acres of winegrapes in Fresno, 600 growers growing more than 42,000 acres of winegrapes in Madera, 1800 growers growing more than 49,000 acres in Napa and 2200 growers growing 64,000 acres of winegrapes in Sonoma.

Sulfur management strategies

Sulfur is a time-tested and staple element in the winegrape grower's toolbox of strategies to prevent powdery mildew. A variety of factors must be considered to effectively manage sulfur use, including weather, canopy management, disease monitoring, appropriate calibration and selection of equipment, appropriate rates and timing of applications, and other interrelated factors. This research focuses specifically on the product choice (formulation) and application rates of sulfur using PUR data for all winegrape growers and fields that had reported pesticide use in each of the four counties considered.

Agricultural sulfur products may be formulated for use as nutrients, soil amendments or pesticides. They are generally finely divided, averaging 93% passage through a no. 325 US sieve, and formulated into either wettable powders or dusts. Wettable powders are formulated by blending dispersants and surfactants together and then milling to a very fine particle size. These products are used primarily as fungicides or miticides. Wettable sulfur can be applied as a ground spray or by aerial application. Dusting sulfur, formulated at 98%, is primarily used as a fungicide, and can also be applied either by ground or air. Dust sulfur products usually cost less than wettable sulfur products.

Most grape growers consider dust sulfur to be more harmful to predatory mites than wettable sulfur[10]. Sulfur dust particles are thought to physically interfere with predatory mites' ability to search for prey, but wettable sulfur particles could have a similar effect[11,26]. Costello and Albers[11] concluded that use rate is the primary determinant of sulfur impact on mite populations. Thus, the differential use rates of dust and wettable formulations, which are dictated by the labels, may lead to differences in their effects on pest mites. This study investigates both formulation types and use rates.

Three sulfur management strategies were defined and investigated using the PUR data: growers who applied only dust sulfur are categorized as 'dust only'; those who applied only wettable, flowable or other forms of liquid sulfur are categorized as 'wettable only'; and those who applied both dust and wettable sulfur are categorized as 'combo'.

PUR data organization and treatment

The PUR database records every pesticide application in California that is reported to local agricultural commissioners. For each pesticide application, there are 32 data fields detailing the product type and active ingredient(s) of the specific pesticide used, the crop, the area over which it was used, the operators who applied it, as well as the date, time, quantity and rate of the application[27]. The spatial resolution of each pesticide application is aggregated to the level of the section, or one square mile area. PUR data collected between 1993 and 2009 were used to delineate the sulfur management strategy for each winegrape grower in the four selected counties. Pesticide use on winegrapes in the selected four counties was queried from the PUR database, and prior to use the data were checked for errors following the error checking procedures developed by Wilhoit et al.[28] to remove outliers from the dataset. Pesticide applications were aggregated by grower, and the sulfur management strategy for each grower was identified by analysis of the dust and wettable sulfur products used. Miticide, insecticide and oil applications for each grower were also identified, with a specific focus on FQPA category I & II ('danger' and 'warning' chemical groups based on the toxicity) miticides and insecticides. Simple statistical methods such as analysis of variance (ANOVA) test with *post-hoc* pairwise *t*-test and Pearson correlation test were used to evaluate the data. Both methods were performed in R[29].

Results and Discussion

Use trends and economic characteristics

Statewide, annual pesticide use on winegrapes decreased by 26% between 1993 and 2009, from 27 to 24 million pounds. Meanwhile, vineyard acreage increased by 73%, from about 366,000 to 632,000 acres. The four counties selected for this study had a more pronounced decrease in pesticide use than the state, with 47, 55, 51 and 36% decreases observed for Fresno, Madera, Napa and Sonoma counties, respectively, over this period (Fig. 2).

Economic impacts on pesticide use decisions may differ between the two distinct wine-growing regions of the north coast and central valley. In Napa and Sonoma counties, grower returns have remained higher than in Fresno and Madera, with the difference growing between 1993 and 2009[30,31]. Higher returns in Napa and Sonoma offer greater flexibility to choose higher cost products and pest management approaches.

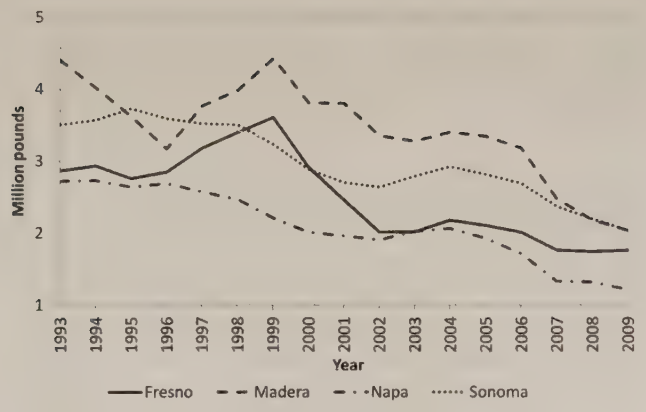

Figure 2. Total pounds of pesticide used in Fresno, Madera, Napa and Sonoma county winegrapes 1993–2009.

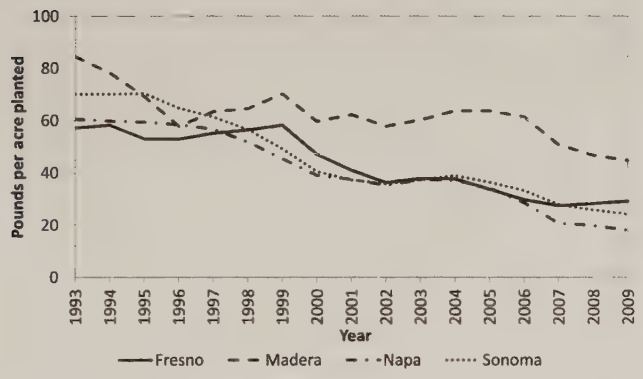

Figure 3. Sulfur use intensity (pounds per acre planted) in Fresno, Madera, Napa and Sonoma county winegrapes from 1993 to 2009.

Sulfur management

Sulfur application rate declined by 56, 49, 64 and 63% in Fresno, Madera, Napa and Sonoma counties, respectively, between 1993 and 2009 (Fig. 3). Sulfur use decline may be related to complaints of drift, odor and air quality problems stemming from sulfur use as development has moved closer to farms. Regulatory entities are also concerned about improving sulfur management to reduce its impact on air quality, which could result in increased regulation in the future. Growers perceive proactive management of sulfur drift as a means of preventing further regulation and ensuring sulfur's ongoing availability as a viable tool to manage powdery mildew in winegrapes[32]. State agencies, such as CDPR, have funded projects to support industry efforts to improve sulfur management[4].

In each of the four counties considered the combination of dust and wettable sulfur formulations was the largest use category in 2000, 2005 and 2009; however, more growers in Napa and Sonoma used only wettable sulfur than in Fresno or Madera (Fig. 4) in all three selected years. In 2000, approximately 38 and 34% of respective Napa and Sonoma growers relied on wettable

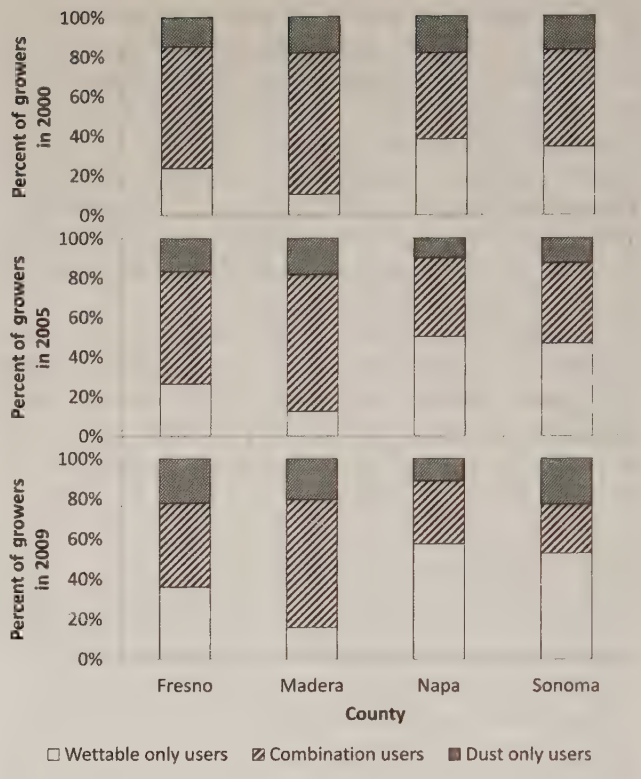

Figure 4. Percent of growers using sulfur dust only, wettable sulfur only or a combination of dust and wettable sulfur in Fresno, Madera, Napa and Sonoma county winegrapes in the year 2000, 2005 and 2009.

formulations only, while only 24 and 10% of respective Fresno and Madera growers relied exclusively on them. In 2005, approximately 50 and 47% of respective Napa and Sonoma growers relied on wettable formulations only, while only 26 and 13% respective Fresno and Madera growers applied them exclusively. In 2009, similar patterns were found for the four counties, 57% for Napa, 53% for Sonoma, 36% for Fresno and 16% for Madera. The percent of growers using only sulfur dust formulations was similar across all counties, ranging from 15 to 19% of growers in 2000, from 10 to 18% in 2005 and from 11 to 23% in 2009 (Fig. 4).

There are several likely reasons why the north coast counties appear to have greater use of wettable sulfur formulations than the valley counties. Cost is probably a factor, with valley counties having much smaller profit margins than coastal counties, making the use of higher-cost wettable sulfur formulations less economically feasible. Another potential reason is the greater role of tourism in the north coast counties. With year-round tourism a major industry in Napa and Sonoma counties, growers there are under greater pressure to avoid unpleasant odors associated with the more drift-prone dust formulations. The higher rainfall and humidity in Napa and Sonoma, relative to Fresno and Madera are also likely factors. Wettable sulfur is an effective treatment once mildew becomes visible. The wetting

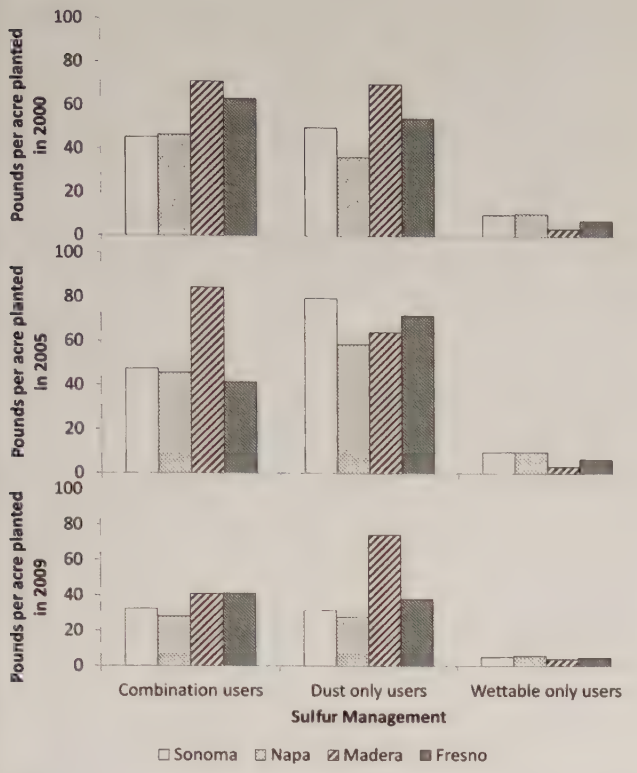

Figure 5. Sulfur use intensity (pounds per acre planted) related to winegrape sulfur management (dust, wettable or combination users) in the year 2000, 2005 and 2009.

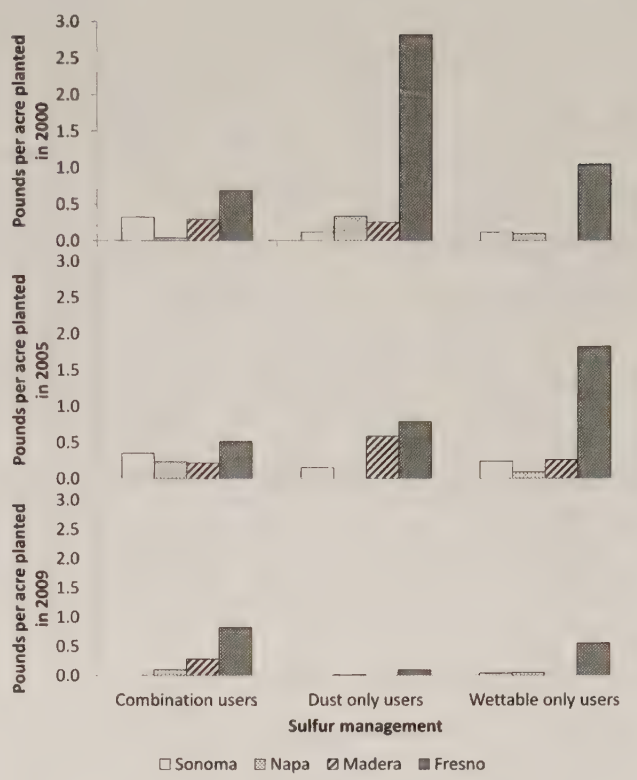

Figure 6. High-risk (FQPA priority I & II) insecticide use intensity (pounds per acre planted) related to winegrape sulfur management in the year 2000, 2005 and 2009.

agents and water eradicate the mildew, while the remaining sulfur kills new spores[33]. This post-infection action of wettable sulfur would be particularly useful in the cooler, damper region represented by Napa and Sonoma, where disease pressure is higher.

Figure 5 and the ANOVA test demonstrate the large statistical difference in use intensity (pounds per acre planted) between sulfur dust or 'combo' and wettable sulfur formulations ($\alpha = 0.05$). However, the results from the pairwise *t*-test showed that there is no significant statistical difference between the use intensities of 'combo' and 'dust only' users ($P < 0.05$). For both of these groups, the sulfur use intensity is lowest in Napa, slightly higher in Sonoma, followed by Fresno, then Madera (Fig. 5).

Insecticide use related to sulfur management

The growers in Madera county who use wettable sulfur only, do not use FQPA I & II insecticides; however, the number of growers using wettable sulfur only in Madera is low (Fig. 6). The growers in Fresno county who apply wettable sulfur only, use FQPA I & II insecticides at 34 and 76% lower use intensity than the combination and dust sulfur users, respectively in 2000, 72 and 35% lower in 2005 and 51 and >700% higher in 2009 (Fig. 6). Based on the ANOVA test, there are no statistically significant differences in the use intensity of FQPA I & II insecticides across sulfur management categories between Sonoma and Napa counties at the significance level of $P < 0.05$. Fresno has the highest use intensity of FQPA I & II insecticides, averaging 1.22, 0.61 and 0.36 pounds per acre planted for 2000, 2005 and 2009, respectively, for all sulfur management categories. This average is 579, 129 and 398% higher in 2000, 2005 and 2009, respectively, than the average use intensity of FQPA I & II insecticides for Sonoma and Napa, which is 0.18, 0.26 and 0.07 pounds per acre planted in 2000, 2005 and 2009, respectively (Fig. 6).

The striking difference in use intensity of FQPA I & II insecticides between the coastal and valley regions did not appear to reflect choices of specific pesticides with different associated use rates. Table 1 shows the top five FQPA I & II insecticides used in Fresno, Madera, Napa and Sonoma counties in 2000, 2005 and 2009 according to total acres treated with the chemicals. When the statewide average use intensities of these top five FQPA insecticides are compared across counties, the differences are not statistically significant (ANOVA test with $\alpha = 0.05$). The statewide average use intensity of the top five FQPA I & II insecticides in 2000 is 2.17, 0.93, 0.16 and 0.58 pounds per acre planted for Fresno, Madera, Napa and Sonoma counties, respectively. In 2005, the use intensity is 1.27, 1.36, 0.77 and 0.83 pounds per acre planted, while the 2009 use intensity is 1.26, 1.11, 0.29 and 0.03 pounds per acre planted for Fresno, Madera, Napa and Sonoma, respectively (Table 1). Therefore, the lower

Table 1. FQPA priority I & II insecticide use ranked according to the top five total acres treated with the chemical in Fresno, Madera, Napa and Sonoma county winegrapes in (a) 2000, (b) 2005 and (c) 2009.

FQPA insecticide	Statewide pounds per acre planted	Fresno	Madera	Napa	Sonoma
(a) 2000					
Fenamiphos	1.75	1	2	4	2
Dimethoate	0.40		3	1	1
Phosmet	1.25	2	1		
Methomyl	0.44	3	5		
Carbaryl	0.77		4	2	3
Diazinon	0.83	4		3	5
Chlorpyrifos	0.83	5			4
Pyrethrins	0.005			5	
Pounds per acre planted	1.04	2.17	0.93	0.16	0.58
(b) 2005					
Fenamiphos	1.50	1	2		
Dimethoate	0.32				2
Phosmet	1.00	4			
Methomyl	0.31	5		3	3
Carbaryl	0.79		3	4	
Diazinon	0.31	3	5	5	
Chlorpyrifos	1.18	2	1	1	1
Pyrethrins	0.02			2	4
Naled	0.50		4		
Endosulfan	0.72				5
Pounds per acre planted	1.11	1.27	1.36	0.77	0.83
(c) 2009					
Fenamiphos	1.07				5
Dimethoate	0.01			5	
Phosmet	1.20	3	2		
Methomyl	0.61		3		
Carbaryl	0.20			4	2
Diazinon	0.62	4			
Chlorpyrifos	1.31	1	1	3	
Pyrethrins	0.01	5		1	1
Malathion	1.50	2		2	4
Naled	0.23				3
Pounds per acre planted	0.81	1.26	1.11	0.29	0.03

use intensity of FQPA I & II insecticides in Napa and Sonoma (Fig. 6) are clearly a function of pest pressure and grower decision-making regarding insect management, and not simply a function of lower label rates based on pesticide choices in each county.

'Dust only' users in Fresno and Madera counties show lower use intensities of FQPA priority III and other insecticides (generally lower risk insecticides) than the 'combo' or 'wettable only' growers (Fig. 7). The choices of applying a combo or wettable sulfur only could be a reflection of grower pest management style. Use of the more expensive wettable sulfur indicates that the grower is aware of the benefits of its use and is willing to adopt the most appropriate material. Wettable sulfur use may be less harmful to predatory mites than sulfur dust, possibly by virtue of lower use rates[11]. Wettable sulfur also has less of an impact on air quality than sulfur dust because it is less prone to drift[34]. However, applying wettable sulfur takes more time than dusting, and complete coverage of all surfaces is critical for suppressing fungal growth. Both dust and wettable sulfur formulations may also contain inert ingredients of concern for volatile organic compound issues[35]. The fact that the 'dust only' users are using less low-risk insecticides supports the assumption that these growers may be less willing to experiment with newer, ecologically more sound materials.

Miticide use related to sulfur management

'Dust only' users in Madera county used 30 and 94% more FQPA I & II miticides than 'combo' and 'wettable only' users, respectively, in 2000 (Fig. 8). However, that trend did not continue in the data for 2005 and 2009. This may be because FQPA review was completed in 2006 and the regulation did affect the practices in the field (Fig. 8).

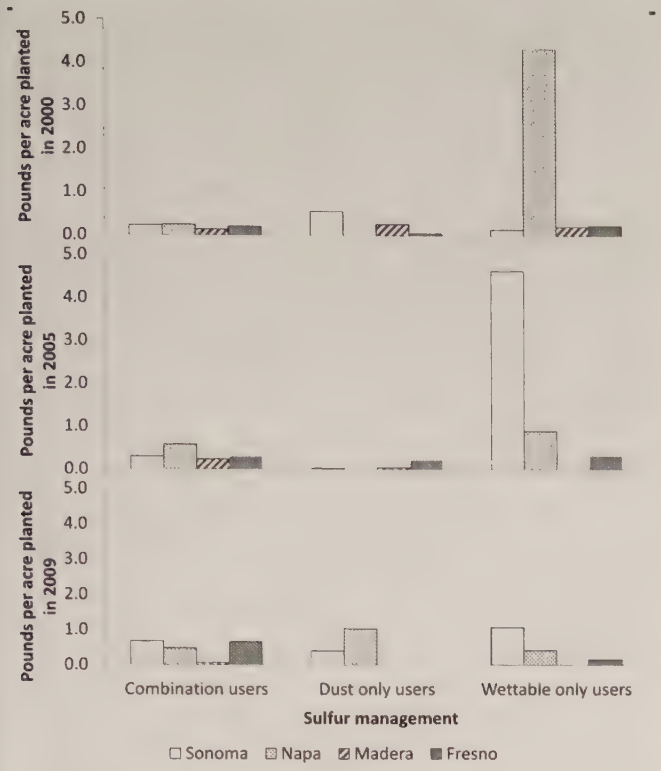

Figure 7. Lower-risk (FQPA priority III and other) insecticide use intensity (pounds per acre planted) related to winegrape sulfur management in Fresno, Madera, Napa and Sonoma counties in 2000, 2005 and 2009.

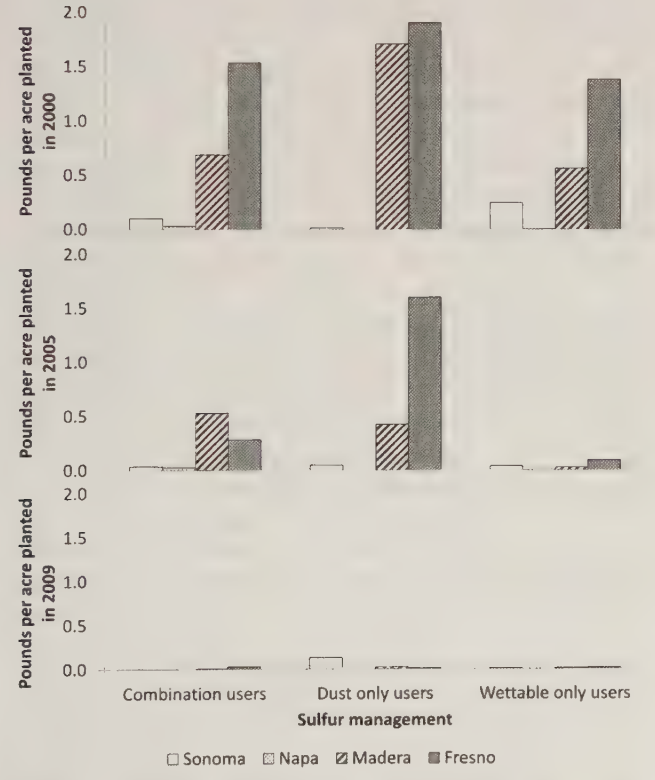

Figure 8. FQPA priority I & II miticide use intensity (pounds per acre planted) related to winegrape sulfur management in the year 2000, 2005 and 2009.

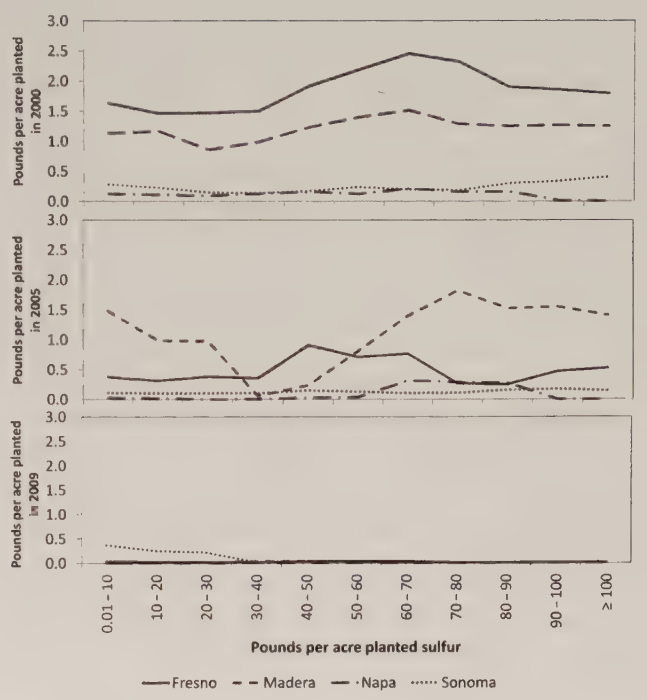

Figure 9. Scatter plot of sulfur use intensity related to FQPA priority I & II miticide use in Fresno, Madera, Napa and Sonoma county winegrapes in the year 2000, 2005 and 2009.

'Dust only' users in Fresno county applied 4 and 12% more FQPA I & II miticides than 'combo' and 'wettable only' users, respectively, in 2000. This trend also did not persist in the data of 2005 and 2009 (Fig. 8). In Napa county, sulfur 'dust only' users used no FQPA I or II miticides in the 3 years we selected. Overall, Napa growers used the least FQPA I & II miticides of all counties considered. The average FQPA I & II miticide use intensity in Fresno and Madera counties was higher than in Napa and Sonoma counties by 1349% in 2000, 1103% in 2005 and 146% in 2009 (Fig. 8).

Some published studies have identified a relationship between sulfur use and increased mite outbreaks in vineyards, and suggest that it is the result of sulfur impacts on either predatory mites[11], the grapevine's pest mite resistance[12] or other factors. Dust from roads and disking exacerbates mite problems and sulfur dust may have a similar effect[22]. The relationship between broad-spectrum insecticide use and mite outbreaks has a similar mechanism of disrupting mite predators. Many pesticide active ingredients are known to be highly toxic to mite predators including sulfur, Carbaryl, Chlorpyrifos, Diazinon, Dimethoate, Esfenvalerate, Malathion, Methomyl, Phosmet, Dicofol and Propargite[18]. Hormoligosis is 'the phenomenon in which subharmful quantities of many stress agents may be helpful when presented to organisms in suboptimal environments'[36]. This mechanism may allow insecticide exposure to increase the reproductive capacity of pest mites, resulting in mite outbreaks after certain pesticide applications.

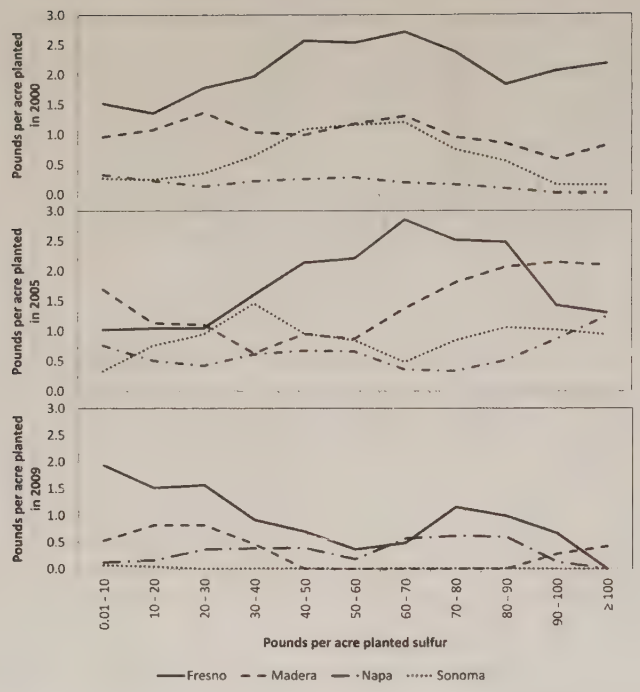

Figure 10. Scatter plot of sulfur use intensity related to FQPA priority I & II insecticide use in the Fresno, Madera, Napa and Sonoma county winegrapes in the year 2000, 2005 and 2009.

Based on the 3 years of data we selected, it is difficult to identify any conclusive trend between higher use intensities of sulfur and higher use intensities of FQPA I & II insecticides and miticides in the four counties (Figs. 9 and 10). Although there may be a slight positive trend observed for Fresno ($r=0.55$, $P=0.08$) and Madera ($r=0.52$, $P=0.08$), the data for Napa and Sonoma did not show a clear relationship. However, Sonoma county data show increased FQPA I & II miticide use intensity when sulfur was used above 20–70 pounds per acre planted in the year 2000 (Fig. 9). The lack of a significant correlation in these two cooler coastal counties could simply be a result of lower overall insect and mite pest pressure. The dramatic differences in use rates of all categories of insecticides and miticides in Napa and Sonoma versus Fresno and Madera across all sulfur use categories (Figs. 6–9) would be consistent with lower pest pressure in Napa and Sonoma. The lack of correlation may also be due to factors other than pest pressure which influence a grower's pest management decisions. These decisions involve a complex set of judgments that will vary based on economic, social, meteorological, seasonal and other complex factors. Thus, pest pressure may not always translate quantitatively into pesticide use reported in the PUR database.

Future research on this phenomenon should continue to focus on various aspects of the relationships between sulfur and broad-spectrum insecticide use and mite outbreaks which have received only limited attention to date. In particular, the mechanisms involving host plant physiology[12] or effects of sulfur (or other fungicides) on naturally occurring mite-pathogenic fungi[13] should be studied in greater detail. Data from the PUR database can be focused more narrowly than county-level totals and averages. A more detailed look at the sulfur use practices of individual combo sulfur users may reveal whether they used predominantly dust or wettable formulations. At the field level, combo users may have treated individual fields with only dust or wettable formulations.

The published correlations between sulfur use and subsequent mite outbreaks are largely based on laboratory and field studies, and often yield conflicting results, ranging from 'no effect'[37] to 'acute toxicity'[15] of sulfur on the same predatory mite species: *G. occidentalis* in these cases. Differences in application levels or formulations of sulfur used, species and life cycle stages studied, timing of exposure and lethality determination, and geographic differences in pest strains likely contribute to these varying experimental results. Here, we have used a novel approach to see if differences in sulfur formulations used by growers under true farming conditions translate into higher annual use of insecticides and miticides reported in the PUR database. The stark differences in sulfur use between the coastal and valley counties which were revealed by the PUR database also underscore the importance of farm economics in governing pesticide choices by growers. This factor should not be ignored when designing alternative pest management strategies. The PUR database has limitations in some aspects of understanding pest management, as it catalogs what the grower did, but not the actual pest conditions in the field. However, coupled with other data sources this resource provides valuable information

for our understanding of pest management history and targeting outreach efforts. The PUR database also makes it possible to identify successful examples of reduced risk farm management practices and promote reduced risk pesticide use in winegrapes and other crops.

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Economic impacts of IPM sampling methods for collards

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Abstract

Integrated pest management (IPM) has been very successful in directing farming to a more environment-friendly production. It is a great tool for producers in transition to organic farming. However, the extent of its economic impact is not well understood by farmers. This study looks at the amount of savings and potential market profitability of using IPM in South Carolina collard production considering alternative scouting methods. Scouting is an essential part of IPM that is usually ignored. This analysis presents a comparative assessment of the merits of conventional sampling (CS) and binomial sequential scouting method (SSM). SSM is a recently developed scouting system for traditionally operated collard farms that is geared toward a more economical execution of scouting without forfeiting the effectiveness of the process. Financial analytical tools, specifically costs and returns methods and sensitivity analysis on prices, were utilized to determine the economic advantages or disadvantages of the two methods. Outcomes indicate that both scouting methods would result in cost savings if used on traditionally operated farms. Particularly, the cost savings per hectare generated from IPM with SSM [3.62% of total cost (TC) and 3.91% of total variable cost (TVC)] is higher than the cost savings from IPM with CS (2.91% of TC and 3.15% of TVC). The difference in cost savings between IPM with CS and IPM with SSM basically came from the less scouting time of SSM that entailed lower labor cost for the farm. Therefore, to attain maximum profitability potential, using IPM with the SSM is a better option. Some may conjecture that the cost savings were insignificant due to the low percentages in cost savings. However, its importance is evident at the potential savings per farm and at the aggregate/state level.

Key words: IPM, scouting/sampling, binomial sequential sampling, profitability, cost savings, collards

Introduction

One of the biggest challenges faced by agriculture is susceptibility to natural occurrences. Pests and plant diseases, which are part of the ecosystem and its natural process, are among the most important constraints in the production of marketable products in general, especially in the production of leafy vegetables such as cabbage, collards and other cole crops[1]. To address this need, many pest management research efforts have been undertaken by the United States Department of Agriculture (USDA), in the areas of biological control, integrated pest management (IPM), invasive species and pesticide application[2]. Of these, pesticide application has been a popular, conventional risk mitigating practice employed by farmers as it has usually been proven to be relatively inexpensive, easy, and effective in meeting market demands for blemish-free crops.

Meanwhile, consumers have become increasingly concerned about healthy lifestyles and the health risks posed by crops grown using synthetic pesticides[3–5]. In response to consumers' rapidly growing demand for healthy food options and producers' inability to employ pesticide-free growing methods without significant yield losses, IPM offers a compromising solution that reconciles the contrasting consumer's concerns and farmers' production goals. Hence, IPM has become a viable alternative and a great tool in transition to organic farming. IPM lowers pesticide application and at the same time controls pest infestations. As the focus of IPM is on prevention, avoidance, monitoring and suppression of pests, chemical pesticides are used only where and when these measures fail to keep pests below damaging levels[6]. Therefore, pest identification and the use of pesticide action thresholds to signal spraying actions is an essential component of IPM.

Information needed on pest identification and pesticide action thresholds are gathered through scouting, making this practice crucial. Scouting with a fixed number of plants to make a decision has been the conventional practice of farms. However, a major criticism by

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producers was that the method was too time-consuming[7]. Thus, more often than not, farmers tend to do it inefficiently or totally skip the process and go directly to spraying. This contention was confirmed by J.P. Smith, a county extension agent of Lexington, South Carolina, based on his interactions with his farmer clientele (April 2007). This defeats the purpose of IPM.

In 2005, Smith and Shepard designed a new way to scout the fields and called the process binomial sequential sampling method (sequential scouting method (SSM))[7,8]. They applied it on collard fields, the most economically important vegetable crop in Lexington County, South Carolina where lepidopteran pests, mainly the diamondback moth and the cabbage looper are major constraints to production. The fixed sample size used by Lexington County farmers in the conventional method for scouting in collard farms is 100 samples. Smith and Shepard found at least 75% reduction in the required samples shifting from conventional sampling (CS) to sequential sampling. The latter method was found to be more time efficient and equally precise as the conventional method. The complete description of the method can be found in the *IPM for Cabbage and Collard: A Grower's Guide*[1]. The study recommendations contend that it will significantly reduce production costs and ultimately increase net benefits to farmers. There has been, however, no published empirical work that can support such an assertion.

This study is designed to address the lack of empirical evidence to establish the efficiency and reliability of the non-conventional scouting method. Specifically, this study will address issues of relative cost efficiency of the SSM *vis-à-vis* the conventional scouting alternative. Financial analytical tools are employed to evaluate comparative cost efficiencies of the two scouting methods on traditionally operated farms. Findings of this research will potentially benefit producers by enhancing their understanding of the financial viability of their farms under the SSM method.

Data and Methodology

Since pesticides are used frequently in collard production and the application cost per acre is high[9,10], scouting makes more sense and somehow can be viewed as a substitute input to pesticides. Especially in the cases where the scouting decision is not to spray, scouting costs displace pesticide costs, which in most of the cases are a better trade-off. This presents a whole new set of potentials in production where the total variable cost (TVC) curve shifts downwards, providing possibilities for bigger profit margins.

To evaluate and compare the effects of CS and SSM under IPM on traditional farms, an experiment using these methods was designed to gather data on the number of labor hours spent, biocontrol or pesticide applications, changes in amount of pesticides used and related items that were further reflected to costs affected. The amount of farm output and other farm-related data were also collected in an effort to replicate and extend the Smith and Shepard study using economic analysis. These records were also used to verify the production and economic data used to develop the Clemson Extension enterprise budget, which is a decision tool popularly used by extension farmer-clients. A schematic diagram explaining the flow of the experiment and comparison of the methods is shown in Figure 1. The diagram also presents information on the number of samples required for each IPM method. The two scouting methods are embedded in the standard production process from seeding to harvesting. Basically, two different methods of pest protection (non-IPM (control) and IPM with CS (IPM-CS)) were compared with SSM (IPM-SSM).The economic comparisons of these methods were kept at the level of whole farm operations in order to account for all relevant and necessary farm costs and determine the overall effect of potential fluctuations in the market price on farm profitability.

The experiment was conducted in one of the biggest collard farms in Lexington County, South Carolina, which is ranked first among the 46 states and among the 3078 US counties in collard production[11]. Two parcels of land measuring around 1.62 ha per parcel were planted with collards and the two different scouting methods were used on each area. The experiment covered an entire cropping season of 16 weeks in 2007.

To verify the claim that affected inputs account for a vital reduction of farm expenses and that the corresponding percentage of savings account for 75% reduction in scouting inputs (samples), the cost-effectiveness of the two scouting methods were analyzed using costs and returns analysis. The Clemson Extension Service enterprise budgets for collards[10] were used in reflecting the changes in costs to gross and net revenues. These enterprise budgets are updated annually to reflect any technological, production changes that farmers should consider and accommodate annual movements in market price volatility. The 2006/2007 collard enterprise budget without any cost changes was used as the costs and returns per hectare for non-IPM farms and, for purposes of this analysis, is designated as the control system (converted from the per acre basis given in the original budget). The changes in costs were particularly reflected on the cost of pesticides (herbicide, insecticide and fungicide), tractor/machinery and the like. The net income per box or profit per box and breakeven price per box were specifically scrutinized in order to determine the required minimum increase in price of collards that would result in maintaining or enhancing farm profits, while employing the scouting activity. Cost savings were also considered, along with short-run production and costs that were used to determine breakeven and shutdown prices for different scenarios. In order to test for the robustness of this study's results, the analysis is extended to include some sensitivity

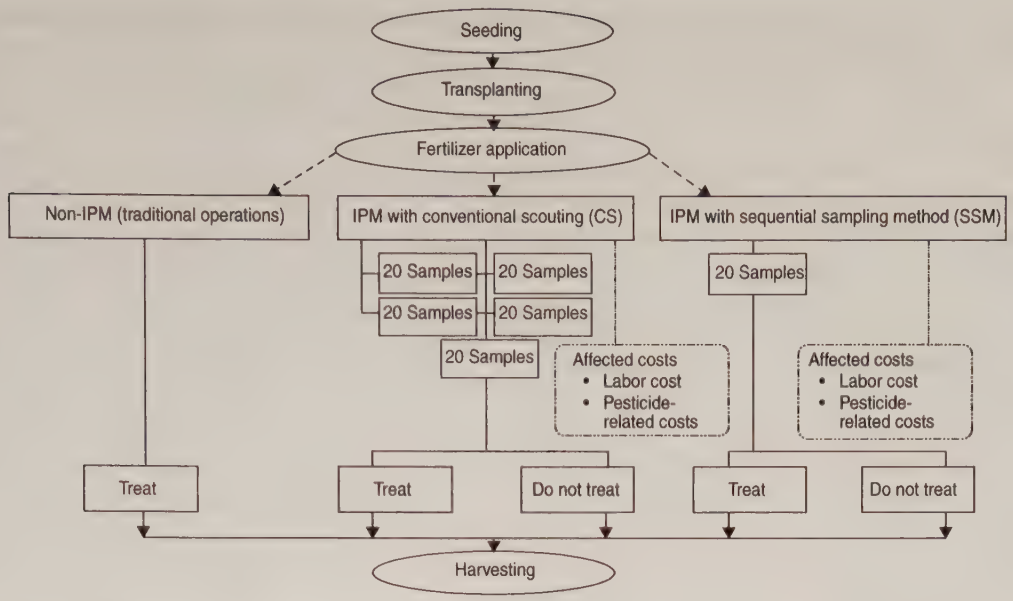

Figure 1. Schematic diagram of the difference in the three methods as a process in planting.

analysis that paid particular attention to changes in farm profit in response to potential input and output price changes.

Results

The first of the three being compared, the non-IPM farms, carry out long-established operations especially when it comes to pesticide management. Over time, farmers subscribed to calendar spraying that protects the crop from pests, but at the same time imposes some negative externalities on the environment. Pesticides have become a substantial part of farm expenses and account for almost 10% of the TVC. Non-IPM farms do not perform any sort of needs assessment or scouting before spraying chemicals on the field. Since most of the operating farms carry out practices under non-IPM circumstances, the costs and returns for this farm were used as the basis of comparison for cost changes due to IPM with conventional practices and IPM with sequential sampling methods.

Alternatively, the two other farm systems fall under IPM and only differ by the scouting method they employed. Under CS, the scout took 100 samples per visit before arriving at a conclusion whether to spray or not to spray pesticides, while sequential sampling only require as few as 20 samples. Every time the scout visits the IPM-CS parcel, he also visits the IPM-SSM plots. On average, it requires 0.45 h or 27 min to obtain 100 samples and 0.16 h or 9.6 min to obtain 20 samples per visit for CS and SSM, respectively. Both methods arrived at the same decision for every visit, verifying the ability of the SSM to be precise with fewer samples and reduced scouting time. For the entire season, the total scouting time under the IPM-CS method took 4.01 h, compared with only 1.48 h that was needed to implement IPM-SSM.

Shifting from conventional to sequential scouting showed a decrease in scouting time. Overall, there is a mean of 59% reduction in time spent per visit, which is smaller than the 75% reduction in samples needed found by Smith and Shepard[7]. However, this measure could be skewed downward due to an outlier found in the data possibly from misreporting of data on the time sheet, such as conducting and reporting SSM results under CS records.

Changes in input costs attributable to conventional scouting

The costs and returns per hectare for a non-IPM farm are displayed in the second column of Table 1. The values under this column are adopted from the enterprise budget from the Clemson Extension Service for 2006/2007. The breakeven prices, i.e., the breakeven price per box for variable costs ($4.96 per box) and breakeven price per box for total costs (TC; $5.84 per box), were the 'base prices' used in comparing the breakeven prices calculated from other farm systems being analyzed.

Most of the input costs differ for an IPM farm using conventional scouting. Its costs and returns per hectare are shown in Table 1 (third column). The variable costs affected were pesticides (herbicide, fungicide and insecticide) and other pest protection costs (beneficial insects), tractor/machinery, labor and interest on operating capital, while general overhead costs changed for the fixed costs. In general, the TC per hectare of producing under IPM with CS was lower than non-IPM farms.

Specifically, the herbicide, fungicide and insecticide costs were smaller due to less application of pesticides brought about by scouting. Different kinds and unit prices of pesticides affected the variation in costs and the use by

Table 1. Collards estimated costs and returns per hectare (irrigated-hand harvest), 2006/2007, 1483 boxes—(around 22.5 kg) harvested in October.

Items*	Non-IPM*	IPM with CS	IPM with SSM
1. Gross receipts			
Collards	11,119.35	11,119.35	11,119.35
Total receipts	11,119.35	11,119.35	11,119.35
2. Variable costs			
Seedlings	711.64	711.64	711.64
Fertilizer			
5–10–10 (spread)	337.44	337.44	334.96
Side dressing-calcium nitrate	187.79	187.79	187.79
Lime (prorated)	129.73	129.73	129.73
Herbicides	12.97	5.83	5.83
Insecticides	369.48	162.47	162.44
Fungicides	282.31	170.55	170.55
Beneficial insects		57.38	57.38
Spreader/sticker	4.94	4.94	4.94
Transplant labor	148.26	148.26	148.26
Irrigation, machinery and labor	184.66	184.66	184.66
Harvest and hauling	1482.58	1482.58	1482.58
Collard boxes	2001.48	2001.48	2001.48
Marketing	741.29	741.29	741.29
Tractor/machinery	156.31	153.13	153.13
Labor	481.91	566.64	510.38
Interest on operating capital	113.64	69.38	69.38
Total variable costs	7346.43	7115.17	7056.41
3. Income above variable costs	3772.92	4004.18	4062.94
4. Fixed costs			
Tractor/machinery	244.63	244.63	244.63
Irrigation	350.04	350.04	350.04
Total fixed costs	594.66	594.66	594.66
5. Other costs			
Land rent	61.77	61.77	61.77
General overhead	649.12	640.38	635.31
Total other costs	710.90	702.15	697.08
6. Total costs	8651.99	8411.98	8348.16
7. Net returns to risk and management	2467.36	2707.36	2771.19
Breakeven yield ($ per ha)			
Variable costs	642.45	642.45	642.45
Total costs	889.55	889.55	889.55
Breakeven price per box			
Variable costs	4.96	4.80	4.76
Total costs	5.84	5.67	5.63

* The list of items and the values under the non-IPM column are adopted from the Clemson Extension Service – Enterprise Budgets, 2006/2007.

farms of beneficial insects in place of other farm chemicals to protect the vegetables from damaging insects. Taken as a whole, the pesticide and pest related costs for the non-IPM farm was $664.76 per hectare, whereas for IPM on a CS farm, the cost was estimated at $396.22 per hectare, which represents a 40% reduction in pesticide costs ($268.54 per hectare). The tractor/machinery expenses were affected because of the reduction in the overall spraying time.

Labor cost is the expense most explicitly affected by the scouting method. The change in labor cost was the net effect of additional hours spent on scouting and smaller labor hours used in spraying. In total 4.01 scouting hours were added and 0.20 spraying hours were deducted from the quantity of labor used for non-IPM farms. To maintain the *ceteris paribus* assumption and maintain consistency with the current situation, the wage per hour used for all of the systems is $9.00 per hour per laborer. Overall, the added cost by labor as a result of scouting in the IPM-CS system was $84.73 per hectare.

Both interest on operating capital and general overhead costs changed due to changes in the input costs discussed

above. First, the interest on operating capital is dependent on borrowed funds financing pre-harvest operation, which comprised costs on seeds, fertilizer and lime, pesticides, machinery repairs, fuel and others. In this case, since the pre-harvest operating cost was smaller in IPM with CS than non-IPM, the interest on operating capital was also smaller. Second, the general overhead cost varies with TVC. Since the net effect of IPM with CS in the system showed a decrease in variable costs, the general overhead cost also decreased as a result. This cost captures all other costs including utilities, telephone and emergencies. The interest rate assumed for both scenarios is 9% as used by the Clemson Extension Service.

Hence, the $44.26 and $20.81 disparity per hectare between non-IPM and IPM with CS for interest on operating capital and general overhead costs, respectively, were indirect effects of the changes in pesticide, tractor/machinery and labor costs.

Changes in input costs attributable to SSM

Costs affected by performing the SSM under IPM were the same as the costs affected by using the IPM with CS techniques. Amounts of changes in the costs of herbicides, fungicides, insecticides, tractor/machinery and interest on operating capital were exactly alike, except for labor and general overhead costs. This was expected because the greater accuracy of results obtained under the SSM *vis-a-vis* the conventional scouting results influences the amount of pest protection-related inputs used in the shortest amount of time.

No additional cost representing the amount related to acquiring the know-how to perform the new SSM was given because the Clemson University Cooperative Extension Service typically does not charge anything for training the farmers. However, in other situations, there might be a training cost to farmers or to supervisors. At this point, the cost associated with training a scout to learn the SSM technique is still unknown. Incorporating this in future analysis as well as accounting for various risk pressures is highly recommended to give a better picture of the scouting effects on the actual farm costs.

Instead of adding 4.01 h spent on scouting conventionally, only 1.48 h of scouting were added to the labor cost under the sequential method. As a result, the expenditure on labor only increased by $28.47 per hectare instead of $84.73 per hectare under the IPM with CS system. In addition, since the TVC were smaller, the general overhead costs decreased to $635.31 per hectare, $5.07 less than the general overhead costs of IPM with CS. Similar components of the overhead costs discussed in the previous section are affected in the same magnitude and direction as those in the IPM-CS system. The decrease in the overhead costs was an indirect effect of the lower labor costs under the IPM-SSM. The costs and returns per hectare of IPM-SSM farms are shown in Table 1 as well (Column 4).

Comparison of the net impacts of CS and SSM relative to non-IPM farms and between CS and SSM systems

After identifying all the input costs affected by performing IPM with CS and SSM in the farm, the overall financial impacts in terms of the amount of savings in costs per hectare were summarized (Table 2). Evidently, there was a positive impact on shifting from non-IPM practices to IPM with scouting—both for conventional and sequential methods. However, it is obvious that the farm under SSM experiences higher savings amounting to $313.40 per hectare (3.62%) compared with a savings of $252.07 per hectare (2.91%) for CS in TC with a difference of $61.33 per hectare (0.73%). The savings per hectare for TVC of $287.53 (3.91%) for SSM and $231.27 (3.15%) for CS were consistent with the savings in TC. The difference in savings in TVC was $56.26 per hectare (0.79%). These percent reductions on savings are not comparable to the 75% reduction in samples needed; however, they could still have significant effects.

Solely considering the percentage changes, the savings do not appear appealing because the percentages seem minute, yet if taken as fraction of commercial production, these could account to significant dollar amounts. The differences in savings between TC and TVC for every system being compared were calculated and can be found in Table 2. The differences between TC and TVC for every scenario were simply the differences in the general overhead costs of the systems in comparison because the sole part of the fixed costs that changed for every system was the general overhead cost causing the variation in savings based on the type of cost.

Whole farm effect and state-level savings

The quantified percentage savings for both methods under IPM seems to be trivial because the per hectare figures are small. For this reason, the whole farm effect on savings was calculated to assess the impact on the grower. Using the average size of about 4 ha for plantings for collards per farm in Lexington County and 2023 ha of land planted with collards in South Carolina, the percentage savings were converted into per farm savings and total savings from collards at the state level (Table 2).

The per farm savings range from around $220 to $1270 while the savings in South Carolina considering that 2023 ha of land are planted with collards range from $114,000 to around $635,000 depending on the method being compared and the type of cost. These numbers provide sufficient evidence to say that the amount of savings derived were significant. It should be noted, however, that these results are limited to a 1-year, 1-farm sample, nevertheless the results of this preliminary step is crucial in realizing and communicating the advantages of adapting the SSM. Reproduction of the study with more sample replications is highly recommended for future work.

Table 2. Savings between different systems by type of cost.

	Savings on TC	Savings on TVC	General overhead cost savings
Non-IPM to IPM-CS			
Percent	2.91	3.15	
Dollar per hectare	252.07	231.27	20.80
Dollar per farm	1020	936	
Dollar savings for state	510,050	467,950	
Non-IPM to IPM-SSM			
Percent	3.62	3.91	
Dollar per hectare	313.40	287.53	25.87
Dollar per farm	1268	1164	
Dollar savings for state	634,150	581,800	
CS to SSM			
Percent	0.73	0.79	
Dollar per hectare	61.33	56.26	5.07
Dollar per farm	248	228	
Dollar savings for state	124,100	113,850	

Table 3. Sensitivity analysis on profit.

		Realized profits					
		Non-IPM profit		IPM-CS		IPM-SSM	
Percent changes in price	Price ($/box)	Based on TC ($/box)	Based on TVC ($/box)	Based on TC ($/box)	Based on TVC ($/box)	Based on TC ($/box)	Based on TVC ($/box)
−20	6.00	0.16	1.04	0.33	1.20	0.37	1.24
−15	6.38	0.54	1.42	0.71	1.58	0.75	1.62
−10	6.75	0.91	1.79	1.08	1.95	1.12	1.99
−5	7.13	1.29	2.17	1.46	2.33	1.50	2.37
0	7.50	1.66	2.54	1.83	2.70	1.87	2.74
5	7.88	2.04	2.92	2.21	3.08	2.25	3.12
10	8.25	2.41	3.29	2.58	3.45	2.62	3.49
15	8.63	2.79	3.67	2.96	3.83	3.00	3.87
20	9.00	3.16	4.04	3.33	4.20	3.37	4.24

Sensitivity analysis

Responses of profit on potential changes in market prices were studied. The profit per box of collards grown under the non-IPM system using the sales price of $7.50 per box were $1.66 per box based on TC and $2.54 per box based on TVC (returns above variable costs). These are the baseline profits to be used as the basis for changes in profitability. Realized profits under the three systems at different price levels were obtained and are shown in Table 3.

At a sales price of $7.50 per box, which is basically the no-change in price scenario, it is evident that the profits under IPM-CS and IPM-SSM were higher than the baseline profits mentioned above. In fact, the profits of the two systems under IPM were all higher than the non-IPM farm profit at all price levels. This shows that IPM improves farm profitability by eliminating unnecessary pesticide costs.

Nevertheless, looking closely at the figures, it can be noted that the profits from IPM with SSM are higher than the profits from IPM with CS at all price levels. The $1.83 profit per box under IPM with CS could be stretched to $1.87 per box under IPM with SSM. Thus, in terms of financial benefits among the three systems in comparison, production under IPM with SSM is better than the other systems. IPM with SSM is the optimal choice for profit maximization.

The least prices needed by the CS and SSM systems that can yield a higher profit and their corresponding percent changes relative to the sales price of traditional farms are calculated. All of the minimum prices required to have a higher profit than the traditional farm are lower than $7.50 per box. This suggests that CS and SSM systems can generate a higher profit than non-IPM produce at lower prices. More importantly, the necessary percent changes for the minimum prices are negative, which accommodate a 2 and 3% decrease in sales price for

IPM-CS and IPM-SSM, respectively, and still generate a higher profit.

Summary and Conclusions

Gradual alteration of the whole farming system to an environment friendly production is a very protracted process. Breakthroughs in farm activities toward the transition away from chemical-based inputs bring agriculture one step closer to its desired goal of being sustainable, which entails not only environmental health but also economic profitability and social and economic equity[12]. Working toward this goal, three pest protection methods were compared in this study, namely non-IPM, IPM-CS and IPM-SSM. Results confirmed that IPM with CS and IPM with SSM are both cost-effective and provide savings to the farm. The latter has higher savings as a result of the reduced scouting time required under SSM, thus lowering labor costs for the farm. Both represent improvements over traditional systems and, thus, offer both positive environmental and economic benefits to the farming business. These production methods can be reliable tools for farms that are in transition to sustainable farming.

More importantly, given the volatility of prices and market conditions, positive and negative shocks on sales price were analyzed to determine the capability of the suggested methods to accommodate possible positive and negative price changes. This analysis contends that both methods under all scenarios analyzed are able to sustain a decrease in sales price of at least 2% greater than traditional systems. This is a robust characteristic and only adds to the quality of the two methods. Matching up IPM with CS and IPM with SSM, the latter was proven to be more cost-effective and considered the option with the best potential to optimize the farm's profit-generation capability.

Subsequent application of the SSM to other production systems, specifically to other brassica crop production in the south, has great potential and is highly recommended. Specifically, this scouting method could be highly beneficial to the production of other leafy greens for instance, turnip greens, mustard greens, kale and other cole crops that are affected by similar pests.

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The phenology, yield and tuber composition of 'early' crop potatoes: A comparison between organic and conventional cultivation systems

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Abstract

Potato production in several Mediterranean countries is focused on the 'early' crop type, and is generally associated with intensive applications of farming inputs. Here we report, for the first time, a comparison of crop performance between organic and conventional cultivation systems. Three cultivars were tested over two seasons (2007 and 2008) to record their phenology, yield and tuber chemical composition. The organic cultivation system was less productive than the conventional one across both years with respect to total yield, but in the season (2008) when late blight infection was not severe the difference was narrowed from 7% (Ditta) to 20% (MN 2-1577 S1). The Italian breeding clones (MN 1404 O5 and MN 2-1577 S1) deserve specific consideration due to their higher total yield and nutritional value (in terms of total protein and vitamin C content) under organic cultivation system than the cultivar Ditta. In addition, the organic farming produced tubers with a lower nitrate content, an important benefit in the context of human health. In conclusion, our results indicate that organic cultivation of 'early' potatoes can deliver acceptable agronomic and qualitative performances. However, the response of the 'early' crop potato to organic farming depends upon both seasonal conditions and cultivar choice. In particular, the selection of appropriate cultivars is one of the key aspects to optimize this environmentally friendly production system.

Key words: 'early' crop potato, cultivation system, organic, conventional, cultivar, season

Introduction

The past few decades have witnessed an increasing interest in environmental safety and food quality, and this has driven a major expansion in the organic farming sector[1]. In contrast to conventional production systems, those followed in the organic sector are popularly perceived as being environmentally sustainable and consumer friendly, since they specifically avoid the use of either synthetic fertilizers or organic pesticides[2–4]. The relative healthiness of organic produce, however, remains controversial[5,6]. Nevertheless, organic production systems have been established all over the world and have expanded considerably in Italy during the 1990s, consolidating its position as the leading European producer of organic food[7]. Potato is one of the crops to be successfully grown organically both in Italy[8,9] and elsewhere[10–12].

Potato cultivation in the Mediterranean Basin occupies about 1 Mha and produces 27 Mt of tubers[13], mostly cropped in the period from March to June to exploit the 'early crop potato' market[14]. In southern Italy (Sicily, Campania and Apulia), potato production contributes significantly to the agricultural economy. 'Early' crop potato production conventionally requires major inputs of inorganic fertilizer and pesticide[14], which can result in undesirable residues in both the tubers[11] and the soil[15]. As a result, organic systems of 'early' crop potato production are on the rise. Despite the price premium available for

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organic potato tubers (over 200–250€ per ton more than conventional ones), there has been to date little effort to breed cultivars specifically for the organic production of 'early' crop potatoes. Parisi et al.[16] have suggested that such cultivars need to show a reliably high yield in a low-input production system, be efficient with respect to nutrient uptake, be able to provide early ground cover rapidly, show a good level of resistance/tolerance to the common biotic and abiotic stresses and a high suitability to low temperatures of storage.

Yield levels are typically lower in organic systems than in conventional high-input ones[9,12], although a few examples have been reported showing that performance has not been compromised by the absence of high inputs[8,10]. Aspects of the chemical composition of main crop tubers have been shown to depend on the cultivation system. Some authors[3,17] have reported that tuber dry matter content is higher in organically grown potatoes than in conventionally grown ones, but Pither and Hall[18] claimed that the opposite was true. Contradictory data have also been published with respect to both vitamin C[10,19,20] and total protein content[5,9,17]. A recent review by Lairon[21] concluded that the nitrate content of organically grown foodstuffs (including potato) is in general lower than in conventionally grown produce. Since potatoes belong to the category of vegetables with low nitrate content, these results might help to reduce the total intake of nitrate, mainly in the countries where potatoes are highly consumed. This is important, since nitrates are thought to act as precursors for carcinogenic nitrosamines in the human gut, and for the nitrites associated with methaemoglobinaemia in infants and the elderly[22]. The tuber composition of organically grown 'early' crop potatoes, which has not been analyzed in any detail to date, may be greatly influenced by the particular environmental conditions associated with 'early' crop potato production, which substantially modify the morphology and phenology of the crop[14]. These tubers are essentially immature, and so differ qualitatively from main crop ones[23]. As a result, little of the literature describing the characteristics of main crop potatoes can be used to make inferences regarding 'early' crop tubers. Here, we set out to generate a body of information regarding the behavior of the organically grown 'early' potato crop. We have compared the performance, on a single farm, of a cultivar much used by organic farmers with that of two Italian pre-releases. Our aim was to identify the influence of the cultivation system on the crop phenology and growth, yield and the chemical composition of the tubers.

Materials and methods

Site, climate and soil

The trials were conducted over two seasons (2007 and 2008) at a commercial farm located on the coastal plain of Siracusa (37°01′N, 15°12′E, 30 m above sea level), which is an area typical for 'early' crop potato cultivation in southern Italy. The local climate consists of mild winters and hot, rainless summers. The soil type is a calcixerollic xerochrepts[24], with pH 7.7 and a soil composition of 48% sand (2–0.02 mm), 18% silt (0.02–0.002 mm), 34% clay (<0.002 mm), 6% limestone, 1.8% organic matter, 0.2% total nitrogen, 0.0028% available P_2O_5 and 0.018% exchangeable K_2O. A layer, 0.25 m thick (from −0.05 to −0.30 m), where about 90% of active roots were located, was considered for the soil analysis. In the 2 years of the trials, we used two adjoining fields in the same area to guarantee pedoclimatic conditions as uniform as possible for both types of cultivation systems. The plots used for organic cultivation converted fully to organic farming 10 years ago by following a period of lemon production with a crop rotation involving potato, zucchini and wheat. The conventional plots (involving the same crop rotation) were separated from the organic ones by rows of trees, resulting in a total distance of 50 m.

Plant material, experimental design and management practices

In each cultivation system (conventional and organic), a randomized block design with three replications repeated over 2 years was used. Each block included three cultivars (Ditta, MN 1404 O5 and MN 2-1577 S1) with a plot size of 42 × 45 m, each containing 101 plants. Ditta is an Austrian cultivar used for organic production in several European countries and is a popular choice for Sicilian organic producers. Its firm, deep yellow colored flesh is important for the export market. The two Italian pre-release clones MN 1404 O5 and MN 2-1577 S1 are currently being assessed by the Italian National List of Varieties; their provisional cultivar names are Gilda and Unidea, respectively.

In both cultivation systems and seasons, disease-free non-pre-sprouted 'seed' tubers, from a single seed lot, were manually planted on January 23, 2007 and January 30, 2008 at 5.33 plants m^{-2}. The conventional cultivation system (CONV) was managed according to local farming practices. Prior to planting, the soil was disinfected with 23 and 4 kg ha^{-1} of fipronil and pencicuron, respectively, to control wireworms (Elateridae) and stem canker (*Rhizoctonia solani* Kuhn). A pre-emergent treatment with flufenacet and metribuzin (0.8 and 0.5 kg ha^{-1}) was included to control weeds, and cymoxanil (11 kg ha^{-1}) was applied to control late blight [*Phytophthora infestans* (Mont.) de Bary], both prophylactically and curatively. Insect infestation (aphids and the Colorado potato beetle) was prevented by spraying with deltamethrin at a dose of 0.5 kg ha^{-1}. The crop management of the organic cultivation system (ORG) followed current EU regulations (Regulation CE 834/2007, 889/2008, 967/2008, 1235/2008 and 1254/2008). No herbicide treatment was given, since the crop was kept free of weeds through

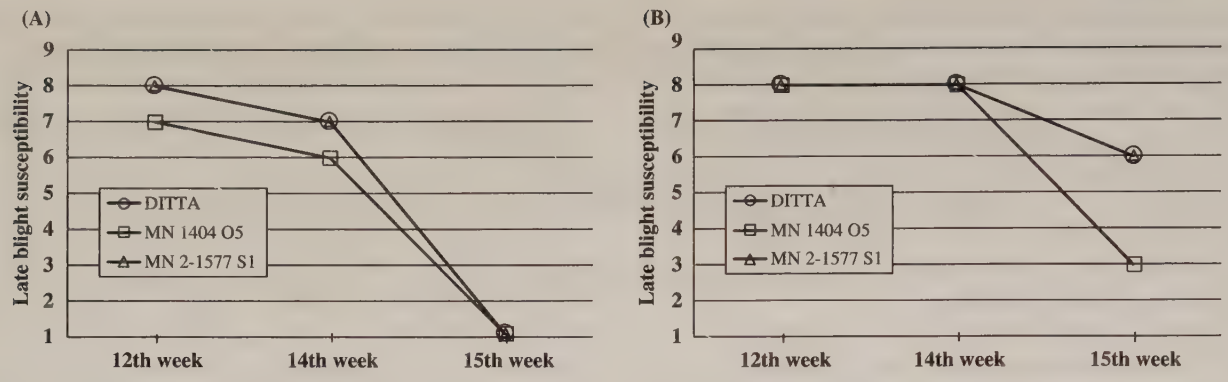

Figure 1. Late blight susceptibility of the three studied potato cultivars under organic cultivation system in (A) 2007 and (B) 2008. Scoring is based on the Malcolmson scale[25], where 1 represents highly susceptible and 9 highly resistant.

hand-hoeing when necessary. Late blight was controlled by applying copper hydroxide and a pitchy extract from propolis (bee glue), which, in both seasons, were applied weekly after plant emergence and throughout the plant development [from 60 to 100 days after planting (DAP)] at a dose of 5 and 3 kg ha^{-1}, respectively.

The same fertilizer regime was applied in CONV and ORG, namely a pre-sowing application of 50 kg ha^{-1} N, 100 kg ha^{-1} P_2O_5 and 120 kg ha^{-1} K_2O in the form of either NPK synthetic fertilizer (CONV) or a mixture of feathers and torrefied bone and meat meal (ORG). A further 83 kg ha^{-1} N was supplied at the tuber induction stage in the form of ammonium nitrate (CONV) or a mixture of dried manure and hydrolyzed pelt (ORG). Drip irrigation was initiated once the accumulated daily evaporation rate had reached 40 mm, and the total water supplied was 128 and 198 mm in 2007 and 2008, respectively. Five irrigation treatments were performed during each season.

Crop phenology, growth, disease and yield

Emergence date was expressed as the number of DAP when a visible shoot had emerged from 80% of the 'seed' tubers, while crop maturity date (in DAP) was defined as when 80% of the plants either showed signs of haulm desiccation or when the tuber periderm could not be removed by hand rubbing. No attempt was made to manually defoliate plants infected with late blight, in order to allow for a proper evaluation of plant susceptibility during the 12th, 14th and 15th weeks after planting, using the Malcolmson scale[25]. The latter measures the intensity of foliar blight caused by *P. infestans* by assessing the overall amount of necrotic tissue per plant on a scale from 1 (highly susceptible) to 9 (highly resistant) (Fig. 1). The number and fresh weight (FW) of shoots were determined at 90 DAP, in both seasons, aimed at providing additional information on crop growth. For the determination of total yield, tubers were harvested manually when about 70% of haulms were fully desiccated, and the number and weight of marketable and unmarketable tubers per plant were determined. Tubers which were greened, misshapen or displayed pathological damage were classed as unmarketable, as well as those with weight lower than 20 g. This allowed the calculation of the number of marketable tubers per hectare, mean tuber weight, total yield and yield losses (expressed as percentage of unmarketable yield of the total yield).

Determination of tuber chemical composition

Tuber chemical composition was analyzed from a representative sample of at least 20 marketable tubers (Φ 35–70 mm), of uniform size and disease-free, per replicate. The tubers were washed, dried with tissue paper, weighed, diced and blended in a domestic food processor at 0°C (Kenwood Multipro, Milan, Italy). Finally, an amount of the resulting slurry was freeze-dried (Christ freeze drier, Osterode am Harz, Germany) and stored at −20°C until analysis of total protein content, another sample was used fresh for vitamin C content determination and the remaining portion was oven-dried at 65°C, until a constant weight was reached, in order to determine the dry matter content. Then, the dried material was ground and passed through a 1-mm sieve, and used for the determination of nitrate content.

Vitamin C was quantified using the 2,6-dichlorophenolindophenol (DCPIP) dye method[26]. An amount (100 g) of slurry was mixed with 50 ml of 8% (v/v) acetic acid and 50 ml of 3% (w/v) metaphosphoric acid, vigorously shaken and filtered. Then, an aliquot (10 ml) of the filtrate was titrated with DCPIP until the development of a rose–pink color. The vitamin C content was expressed as mg kg^{-1} of FW.

Total protein content was determined according to Bradford[27] assay. Briefly, 20 mg of powdered freeze-dried material was suspended in 5 ml with NaOH (0.5 N), as described by Snyder and Desborough[28], vortexed and centrifuged (3000 *g*, 10 min) at 25°C. Then, an aliquot of extract (0.3 ml) was transferred into plastic cuvettes and 2.8 ml Bradford reagent were added. The solution was thoroughly mixed by inversion, and after being held

Table 1. Local mean maximum and minimum temperatures and rainfall over the period 1959–1988, and in 2007 and 2008.

Meteorological variables	Year	Month			
		February	March	April	May
Max. air temperature (°C)	1959–1988	15.6	17.0	19.8	24.7
	2007	19.8	21.5	20.9	22.6
	2008	12.6	17.8	20.1	22.8
Min. air temperature (°C)	1959–1988	8.0	8.9	10.7	14.1
	2007	10.4	12.6	13.0	16.4
	2008	8.5	11.2	12.3	13.7
Rainfall (mm)	1959–1988	41.9	38.8	27.9	11.5
	2007	25.5	122.9	19.7	2.2
	2008	44.8	18.1	36.5	11.0

for 10 min at room temperature, the absorbance was measured at 595 nm using a Shimadzu 1601 UV–Visible spectrometer (Shimadzu Corp., Tokyo, Japan). The total protein content was determined on the basis of a standard calibration curve generated with known amounts of bovine serum albumin (BSA) standard. The results were converted on FW basis, considering the dry matter content, and expressed as $g\,kg^{-1}$.

Nitrate content was determined using an ion-selective electrode method[29]. A 5 g sample of powdered dried material was homogenized in 100 ml 0.04 M $(NH_4)_2SO_4$ for 30 min on a magnetic stirrer. The ionic strength of the solution was directly measured with a pH meter equipped with an ion selective electrode (Jenway, Essex, England). The results were converted on FW basis, considering the dry matter content, and expressed as $mg\,kg^{-1}$.

All reagents were purchased from Sigma-Aldrich (Milan, Italy), and solutions were prepared in double-distilled water. All analyses were performed in triplicate.

Statistical analysis

Bartlett's test was used to test for homoscedasticity, following which the data were subjected to a two-way analysis of variance (ANOVA), considering the cultivation system and cultivar as fixed factors and the growing season and blocks as random factors[30]. Means were separated by Tukey's HSD test, when the *F*-test was significant. Percent values were transformed to arcsin $\sqrt{x}$ (Bliss transformation) prior to analysis and then subjected to ANOVA; untransformed data were reported and discussed.

Weather conditions

Rainfall during the 2007 growing season was above average (170 mm versus a 30-year mean of 120 mm), as were the mean maximum (21.2 versus 19.3°C) and minimum (13.1 versus 10.4°C) temperatures in the period from February to May. The 2008 growing season was more typical as compared to the long-term climate, experiencing about 60 mm less rain than in 2007, and a lower than average mean temperature during the growing period (Table 1).

Results and discussion

Crop phenology and growth

Unlike the emergence date the crop maturity date was significantly affected by cultivation system, but its effects were season dependent (Table 2). Passing from 2007 to 2008, crop maturity date was markedly delayed, especially under ORG (14 days) (Fig. 2). In contrast, no uniform trend was observed between cultivation systems within a growing season (Fig. 2). It is difficult to ascertain which possible mechanism was involved, since no literature data are available about the relationship between the 'early' potato crop phenology and kind of cultivation system.

No significant effect on crop growth, as measured by number and FW of shoots per plant, was shown in relation to cultivation system (Table 2). This, probably, implies that nutrient availability under ORG does not represent a limit for the regular crop growth.

Tuber yield and its components

The effect of the cultivation system on the total yield was both cultivar- and season-dependent (Fig. 3). In both seasons, apart from MN 1404 O5 in 2007, the total yield was significantly lower under ORG (Fig. 3). However, the relative loss in total yield under ORG was comparable to that experienced both in main crop potato[9] and in other crops[31]. Yield reduction suffered under low-input systems may be ascribed to a limitation in the amount of available soil nitrogen[32] and to a less complete and slower control of diseases[8]. In the latter view, under ORG all the studied cultivars produced a greater total yield in 2008 than in 2007, mostly thanks to their lower late blight susceptibility recorded in the second season (Fig. 1). The latter was, however, characterized by less severe late blight attacks (data not shown). In particular, the yield of ORG-grown Ditta was 150% higher in 2008 than in 2007 (Fig. 3). The reduced disease pressure experienced in 2008 also resulted

Table 2. *P*-values, resulting from ANOVA, for all the studied variables.

Source of variation	Cultivation system (CS)	Cultivar (C)	CS×C	CS×Se[3]	C×Se	CS×C×Se
Degrees of freedom	1	2	2	1	2	2
Variable						
Emergence date (DAP)[1]	0.48	0.10	0.20	0.07	0.97	0.23
Crop maturity date (DAP)	0.89	0.98	0.40	<0.0001	<0.0001	0.10
Number of shoots per plant	0.93	0.35	0.45	0.18	0.06	0.16
Mean shoot FW[2] (g)	0.24	0.56	0.24	0.39	0.51	0.68
Number of marketable tubers per hectare ($\times 10^4$)	0.12	0.84	0.62	0.001	<0.0001	<0.0001
Mean tuber weight (g)	0.51	0.12	0.30	<0.0001	<0.0001	<0.0001
Total yield ($t\,ha^{-1}$)	0.12	0.18	0.24	0.02	<0.0001	<0.0001
Yield losses (% of total yield)	0.05	0.14	0.30	0.51	<0.0001	<0.0001
Dry matter ($g\,kg^{-1}$ FW)	0.87	0.41	0.25	0.005	0.0003	0.006
Total protein ($g\,kg^{-1}$ FW)	0.17	0.75	0.02	0.22	0.01	0.10
Vitamin C ($mg\,kg^{-1}$ FW)	0.36	0.47	0.006	0.006	0.66	0.30
Nitrate ($mg\,kg^{-1}$ FW)	0.29	0.12	0.07	0.01	0.0006	0.26

[1] DAP, days after planting.
[2] FW, fresh weight.
[3] Se, season.

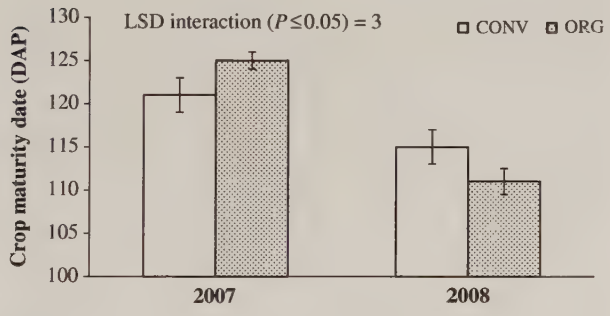

Figure 2. Crop maturity date of 'early' potato crop as affected by the interaction between cultivation system and season. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system; DAP, days after planting.

in a lower yield differential between ORG and CONV, narrowing from 7% (Ditta) to 20% (MN 2-1577S1) (Fig. 3). It is well known that the yield reduction due to premature destruction of foliage and diseased tubers depends on timing and severity of late blight infection[33]. Olanya et al.[34] noted that the level of damage by *P. infestans* is directly related to the environmental conditions prevailing over the growing season of the crop. *P. infestans* infection and spread are facilitated by prolonged periods of leaf wetness and high humidity[35], as it was in 2007. The latter was characterized by higher mean temperatures and total rainfall (especially at the end of March) that created the optimal conditions for the following development of late blight infection. In Figure 3, the yield losses, expressed as the percentage of unmarketable yield of the total yield, are also shown. The effect of cultivation system on this variable was both cultivar- and season-dependent (Table 2). In 2007,

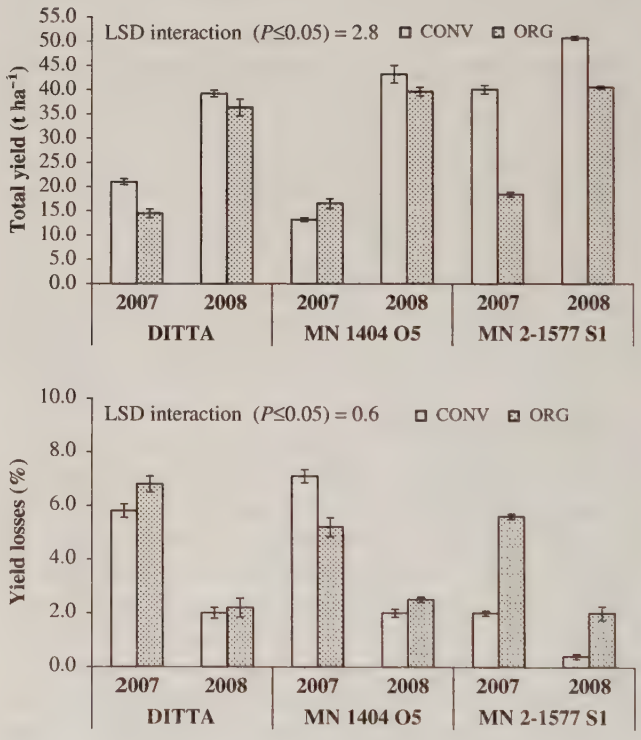

Figure 3. Total yield and yield losses of 'early' potato crop as affected by the three-way interaction between cultivation system, cultivar and season. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system.

passing from CONV to ORG, yield losses increased in Ditta and MN 2-1577 S1, whereas they decreased in MN 1404 O5 (Fig. 3). In contrast, in 2008 higher yield losses under ORG were only noted in MN 2-1577 S1 (Fig. 3). Just as for total yield, the number of marketable tubers per

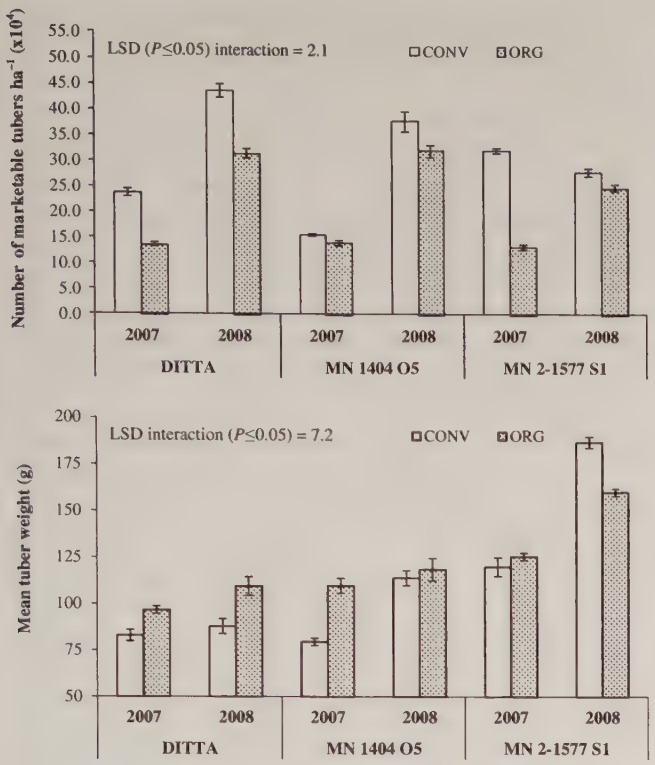

Figure 4. Yield components of 'early' potato crop as affected by the three-way interaction between cultivation system, cultivar and season. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system.

hectare and mean tuber weight were significantly affected by 'cultivation system × cultivar × season' interaction (Table 2). In particular, the number of marketable tubers per hectare of ORG-grown Ditta and MN 1404 O5 was, on average, 130% higher in 2008 than in 2007 (Fig. 4). This trend toward a better productive performance of the 'early' potato crop in 2008 was also shown for the mean tuber weight, except for CONV-grown Ditta tubers (Fig. 4). In particular, passing from 2007 to 2008, the mean tuber weight of MN 2-1577 S1 rose from 126 to 160 g under ORG and from 120 to 187 g under CONV (Fig. 4). As suggested by Warman[36], variation in weather has a greater influence on productivity than the kind of fertilizer adopted. The lower rainfall and mean temperature experienced in 2008 (Table 1) limited the damage caused by late blight infection, and this was probably largely responsible for the better performance of the crop in that year.

Chemical composition of the tuber

Tuber chemical composition is thought to be more strongly influenced by the choice of cultivar and location than by the cultivation system[17]. In our study, the cultivation system significantly affected the chemical variables under study, with the extent of the effect being season and/or cultivar dependent (Table 2). In

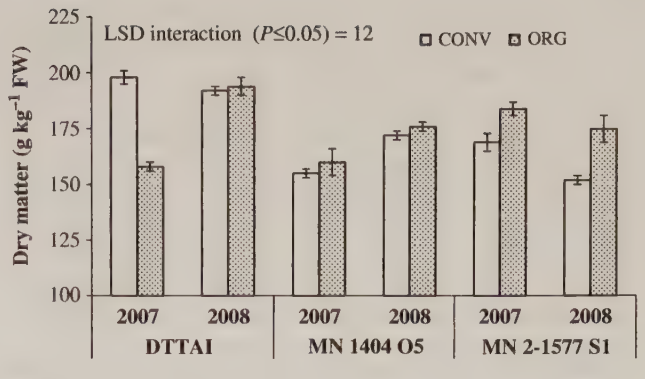

Figure 5. Dry matter content of 'early' potato tubers as affected by the three-way interaction between cultivation system, cultivar and season. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system.

particular, there were significant differences between ORG and CONV with respect to the dry matter content accumulated by cultivars and between seasons (Fig. 5). In this sense, in both seasons MN 2-1577 S1 was more efficient in ORG than in CONV, a trait that is important in the context of cooking quality[37]. In contrast, the dry matter content of MN 1404 O5 was unaffected by the cultivation system in both seasons (Fig. 5). This implies that combination of cultivar and cultivation system may lead to a product with fairly distinct nutritional traits.

Cultivar, growing environment, harvesting technique and post-harvest storage conditions are all important for the determination of tuber vitamin C content[38]. Here, this parameter was significantly affected by the cultivation system, but its effect was both cultivar- and season-dependent (Table 2). In 2007, the vitamin C content in the CONV-grown tubers was 45% higher than in the ORG-grown ones, whereas no differences were shown in 2008 (Fig. 6). This result does not agree with previous findings[18,20]. However, a similar response to cultivation system has been observed for carrot[39] and tomato[40]. Although plants do not require light to synthesize vitamin C, the amount of light available has an indirect impact on its synthesis through its influence on the photosynthetic production of sugar[41]. According to Toor et al.[42], the higher amount of the soil nitrogen available to the plant (which is more abundant under CONV than under ORG) may have promoted leaf growth, and thus enhanced the photosynthetic rate and hence the production of the sugars needed for vitamin C synthesis. We suggest that this explains the better performance under CONV than under ORG of MN 2-1577 S1 and Ditta, but note that the vitamin C content of MN 1404 O5 did not significantly differ between CONV and ORG (Fig. 7). The carbohydrate pool is not the sole regulator of vitamin C synthesis[43], and other factors therefore probably underlie the extensive genetic variation for vitamin C content observed both here (that of the two MN clones was, on

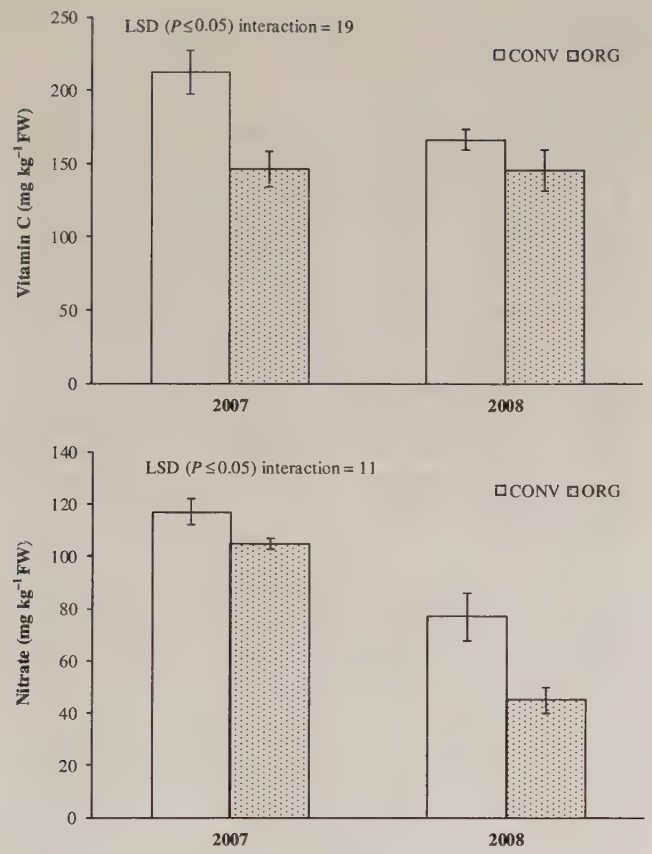

Figure 6. Vitamin C and nitrate content of 'early' potato tubers as affected by 'cultivation system × season' interaction between cultivation system, cultivar and season. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system; FW, fresh weight.

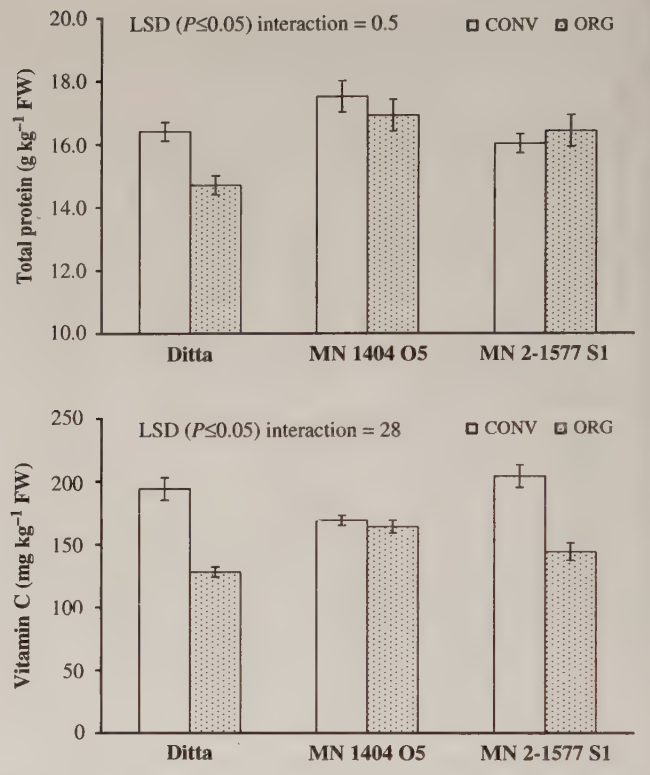

Figure 7. Total protein and vitamin C content of 'early' potato tubers as affected by 'cultivation system × cultivar' interaction. Vertical bars represent the standard error. CONV, conventional cultivation system; ORG, organic cultivation system; FW, fresh weight.

average, significantly higher than that of Ditta) and by Leo et al.[44].

The effect of cultivation system on the total protein and the nitrate content of the tuber was cultivar- and season-dependent, respectively (Table 2). As in main potato crops[3,5], CONV-grown tubers tended to accumulate more nitrogenous substances than ORG-grown ones, due to the higher availability of soil nitrogen under CONV. However, MN 2-1577 S1 had a relatively stable total protein content between the two cultivation systems (Fig. 7). Although the total protein content of ORG-grown tubers was significantly lower than that in CONV-grown ones, it is believed, on the basis of peptide composition, that protein quality (in terms of its nutritional value) is superior in ORG-grown produce[45]. This is an important observation, since in the developing countries the potato, likewise wheat, rice and maize, represents the major source of proteins[46]. In addition, in both seasons the ORG-grown tubers tended to accumulate less nitrate amount compared to the CONV-grown ones (Fig. 6). It is known as organic fertilizers with slowly or moderately available nitrogen (especially composts) compared with chemical ones lead to lower nitrate accumulations in vegetables[21]. No sample from both cultivation systems exceeded, however, the nitrate limit of 200 mg kg^{-1} FW proposed by some European countries (e.g., Germany), which is an important benefit in the context of human health[22]. Finally, the nitrate concentration of CONV- and ORG-grown tubers fell by 34 and 57%, respectively, from 2007 to 2008 (Fig. 6). The differences in nitrate accumulation reflected the contrasting weather conditions between the two seasons (Table 1), which is known to affect the nitrate assimilation rate in plant tissues[47].

Conclusion

Our results demonstrated that 'early' potato crops can be successfully grown under an organic production system. In fact, the latter was able to generate an acceptable decrease of the level of total yield in comparison with the conventional cultivation system, except when heavily attacked by late blight. However, the response of the 'early' crop potato to organic farming depended upon cultivar choice and seasonal conditions. In this view, the Italian breeding clones MN 1404 O5 and MN 2-1577 S1 may be a valuable alternative to the popular cultivar Ditta for 'early' potato production under organic cultivation systems. In particular, they deserve

specific consideration due to their higher yield potential than Ditta under organic farming, as reported especially in the second season. Given the worldwide importance of late blight to potato production, and the detrimental impact of fungicides on the environment, the low susceptibility of the two MN-clones to late blight attacks, at least until the 14th week, deserves specific consideration for the next potato improvement programs. In addition, regardless of season, the ORG-grown tubers of the MN-clones achieved higher total protein and vitamin C content than those of Ditta. Overall, ORG-grown tubers accumulated significantly less nitrate than CONV-grown ones, a finding that has potentially important consequences where potato forms a major component of the diet (e.g., in Scandinavia and Central Europe).

The results here obtained are important considering that a number of EU-based breeding companies have recently released cultivars (e.g., Bionica, Lady Balfour, Raja, Sarpo Mira, Stirling and Toluca) targeted at the organic farming sector, but their yield potential for 'early' crop potato production is uncertain. However, the positive characteristics of the MN clones have to be confirmed further in the context of ORG-grown 'early' crop potato. The development of an integrated management system for organically produced 'early' crop potato requires the breeding of cultivars specifically adapted to low-fertilizer and zero-pesticide systems. At the same time, these cultivars also require a high level of genetic resistance to disease and an acceptable nutritional value. When these cultivars become available, their crop management will have to include the right mix of agronomy, soil fertility and disease (especially late blight) control. These considerations underline the necessity that any enhancement of the yield and quality of organic crops will need a far-reaching optimization of production systems.

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Nitrogen contribution of rye–hairy vetch cover crop mixtures to organically grown sweet corn

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Abstract

Organic cropping systems that utilize winter grown cereal–legume cover crop mixtures can increase plant available nitrogen (N) to a subsequent cash crop, but the rate of N release is uncertain due to variations in residue composition and environmental conditions. A study was conducted to evaluate N availability from rye (*Secale cereale* L.)–hairy vetch (*Vicia villosa* Roth) cover crop mixtures and to measure the response of organically grown sweet corn (*Zea mays* L.) to N provided by cover crop mixtures. Nitrogen availability from pure rye, pure hairy vetch, and rye–vetch mixtures was estimated using laboratory incubation with controlled temperature and soil moisture. Sweet corn N response was determined in a 2-year field experiment in western Washington with three cover crop treatments as main plots (50:50 rye–vetch seed mixture planted mid September, planted early October, and none) and four feather meal N rates as subplots (0, 56, 112 and 168 kg available N ha^{-1}). Pure hairy vetch and a 75% rye–25% hairy vetch biomass mixture ($R^{75}V^{25}$) released similar amounts of N over 70 days in the laboratory incubation. But, the initial release of N from the ($R^{75}V^{25}$) treatment was nearly 70% lower, which may result in N release that is better timed with crop uptake. Cover crops in the field were dominated by rye and contained 34–76 kg ha^{-1} total N with C:N ranging from 18 to 27. Although time of planting and management of cover crop quality improved N uptake in sweet corn, cover crops provided only supplemental plant available N in this system.

Key words: C:N, cover crops, growing degree days, hairy vetch, nitrogen mineralization, nitrogen uptake, rye, sweet corn yield

Introduction

Cover crops can provide a sustainable, on-farm source of N for organic farming systems, reducing the need for off-farm inputs. Crop yields can be maintained by replacing or supplementing N fertilizer with residue N from legume cover crops that fix N, cereal cover crops that scavenge residual available N, or a combination of cereal–legume cover crop residues[1,2]. Effective cover crop N contribution depends on sufficient N accumulation in the cover crop biomass and timely mineralization of the accumulated N[3,4]. The degree of synchrony between N supply from cover crop residue and crop N uptake can influence the efficiency of crop N use[4]. This depends on cover crop composition, environmental conditions, the timing of field events and the type of crop grown[5].

Mixing cereal cover crops with legumes can enhance N cycling because the combined cover crops often have larger biomass and total N content than legumes[1]. Cereal rye is a standard winter cover crop in many regions because it establishes quickly and thrives in cool conditions[6]. Hairy vetch has been shown to be a hardy winter legume, but it prefers warm soil temperatures and therefore establishes more slowly in cool conditions[7]. There is evidence that cover crops grown in biculture have a wider range of tolerance to adverse environmental conditions than grown in monoculture[8,9]. In addition, rye may act as a nurse crop to the hairy vetch, as has been observed in other experiments[10,11].

Cereal–legume mixtures have lower C:N ratios and more plant available N than cereals grown in monoculture[12,13]. Despite these potential benefits it is still unclear as to how the proportions of cereal and legume residues in cover crop mixtures affect N release. Table 1 compares cover crop above-ground biomass, C:N ratios, N uptake and concentration from rye–legume winter cover crop studies conducted across a broad geographical range in the USA. The studies are listed in approximate

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Table 1. Comparison of above-ground biomass and nitrogen from cereal rye–legume cover crop mixtures. All data are for hairy vetch as legume except for Brennan et al., where mixed legumes were used.

	State	Duration (years)	Plant	Terminate	Biomass ($Mg\,ha^{-1}$)	N uptake ($kg\,ha^{-1}$)	Tissue N ($g\,kg^{-1}$)	C:N	GDD	% Vetch biomass	Growth stage
Teasdale et al.[4]	MD	3	Mid September	Mid May–June	9.0–11.2	249–259	23–28				
Brennan et al.[14]	CA	2	Early November	Early April	8.1–14.6	175–185		18–27	1921–2005		Flower
Brennan et al.[14]	CA	2	Early November	Early April	4.7–9.3	115–155		16–27	1955–2057		Flower
Sainju et al.[1]	GA	3	October–November	April	5.7–8.2	84–310	14–38	10–32			
Ranells and Wagger[12]	NC	2	October	Mid April	3.0–5.4	82–200	27–37	14–21		44	R-head V- early bloom
Cline and Silvernail[15 1]	KY	3	Mid September	Mid May	4.2–4.4	90–140	20–33			11–86	R-flower V-early bloom
Griffin et al.[3 2]	ME	3	Mid August	Late May	2.1–4.5	46–170	21–30			< 10	
Lawson[16]	WA	5	Mid September	Late April	4.0–6.0	86–124	19–32	15–25			
This study	*WA*	2	*Mid September*	*Late April*	*3.1–4.8*	*70–76*	*16–22*	*19–27*	*843–1023*	*5–22*	*R-boot V-veg*
Kuo and Jellum[17]	WA	4	Late September	Mid April	1.7–2.4	40–57	18–26	14–21			R-boot V-veg

[1] Cline and Silvernail saw difference in hairy vetch % depending on whether cash crop was fertilized the previous season. We did not see the same effect.
[2] Griffin et al. reported hairy vetch % was less than 10% for one year, but did not report the other 2 years.
GDD, growing degree days; R, rye; V, vetch.

order of cover crop N uptake. Management practices that affect N content and release dynamics from plant residue are changes in residue quality due to planting date, enhanced mineralization due to a combination of cover crops and fertilizer, and a reduction in residue size[18,19,20].

The biological processes involved in soil N transformations are temperature-dependent[21]. In general, temperatures between 20 and 30°C are considered ideal for N mineralization[22]. The effect of temperature on N mineralization can be standardized using a Q(10) factor, which calculates the increase in mineralization rate for every 10°C increase. Research has demonstrated that N mineralization is also a function of growing degree days when moisture is adequate[23]. The advantage of using growing degree days to account for the effect of temperature on N mineralization is that it allows for comparison between laboratory and field trials[24]. Some studies have demonstrated that N mineralization rates increase exponentially with temperature up to 35°C[25]. Others have shown linear correlations for rates of N mineralization[26]. Soil properties and management practices likely account for differences between these results[27,28].

Cover crops influence subsequent crop yield primarily via their effect on N availability[29,30]. Crop yields can be maintained by replacing or supplementing N fertilizer applications with winter legume or cereal–legume cover crop residues[1,2]. Reported N fertilizer replacement values for winter-grown cover crops to subsequent cash crops vary greatly. Several studies examine sweet corn performance after legume and mixed cereal–legume cover crops because of its marked response to N additions. Other researchers have reported yields equivalent to N-fertilizer treatments when grown in rotation with hairy vetch[15,31]. In contrast, some studies have demonstrated poor yield response to rye–hairy vetch blends and lowered N contributions from hairy vetch and rye–hairy vetch blends ($62–75\,kg\,N\,ha^{-1}$) below recommended fertilizer rates[32].

Predicting how much N cover crop residues will supply to subsequent cash crops using field experiments alone is challenging. Nitrogen mineralization is controlled by many factors, including residue composition[12,33] and environmental parameters such as temperature and soil moisture[24,26]. Laboratory incubations that are conducted at optimum moisture and temperature provide a method for comparing N availability across a range of residue types and compositions, which improves our capacity to predict cover crop N availability[34]. Variations in cereal: legume cover crop biomass ratios that can differ widely in the field, can be easily examined using this method. Field data can then be used to validate laboratory results and standardized using growing degree days.

In this study, field and laboratory experiments were conducted to improve our understanding of the N contribution of rye–hairy vetch cover crop mixtures in organic farming systems. This experiment was designed to determine whether rye–hairy vetch cover crop quality

Table 2. Cover crop (R = rye, V = vetch) biomass and total N added, tissue N concentration, C:N ratio, dry matter composition and fiber content[1] (NDF, ADF and ADL) of residues used for incubation.

Cover crop treatment[2]	Biomass ($g\,kg^{-1}$)	Total N ($mg\,kg^{-1}$)	N concentration ($g\,kg^{-1}$)	C:N ratio	Dry matter composition[3] Rye (%)	Vetch (%)	NDF	ADF	ADL
							------- $g\,kg^{-1}$ -------		
V^{100}	1.11	398 b	36 a	13 a	0‡	100	392 b	303 a	63 a
$R^{50}V^{50}$	1.90	549 a	29 b	18 b	50	50	418 ab	264 b	37 b
$R^{75}V^{25}$	1.90	440 b	24 c	21 c	75	25	438 a	253 b	34 b
R^{100}	1.80	293 c	16 d	26 d	100	0	457 a	253 b	21 b

[1] NDF, neutral detergent fiber; ADF, acid detergent fiber; ADL, acid detergent lignin.
[2] ANOVA was run in SAS Proc Mixed. Values within a column followed by the same letter are not significantly different at $P \leq 0.05$.
[3] Dry matter composition in incubation study is percent of each cover crop applied to incubation vessel. Vetch and rye for V^{100} and R^{100} were harvested from pure vetch and pure rye stands in April 2009, while vetch and rye for the $R^{50}V^{50}$ and $R^{75}V^{25}$ treatments were separated from mixed stands and reconstituted as the mixtures noted in the table.

(C:N ratio, fiber fractions) could be managed to increase and time N mineralization from cover crops with nitrogen uptake in organically grown sweet corn with and without feather meal. This research provides information to growers with regard to N credits derived from rye–hairy vetch cover crop mixtures in the Pacific Northwest, USA.

Methods

Laboratory incubation

The release of plant available inorganic N ($NH_4^+ + NO_3^-$)–N from cover crop residues was estimated via laboratory incubation. The experiment included four cover crop treatments and an unamended control. Treatments were labeled based on the percent vetch relative to rye in cover crop biomass: 100% vetch (V^{100}), 50% vetch ($R^{50}V^{50}$), 25% vetch ($R^{75}V^{25}$), and 0% vetch (R^{100}). Application rates for the cover crop residues in the incubations varied among treatments, but were proportional to a 5-year average of cover crop biomass (Table 2) observed for pure rye ($1.80\,g\,kg^{-1}$), pure vetch ($1.11\,g\,kg^{-1}$), and rye-vetch biomass mixtures ($R^{50}V^{50}$; $1.90\,g\,kg^{-1}$), ($R^{75}V^{25}$; $1.90\,g\,kg^{-1}$) measured during a previous 5-year study located on the same field site at WSU Puyallup, WA, USA[16].

Cover crop samples used in the incubations were harvested from stands on a field unit adjacent to the field experiment planted to sweet corn (described in the 'Experimental design' section). This field unit was located on the same soil series as the field experiment and was planted and managed using identical seed sources and management practices as those of the field experiment. The cover crops were planted in mid September and harvested in late April. Above-ground cover crop biomass was harvested from a 6.1×0.91 m swath approximately 5 cm above the soil surface with a flail mower. The harvested forage from each plot was weighed wet, and a subsample (approximately 500 g) was collected and oven dried at 55°C to determine dry matter. Hairy vetch and rye for V^{100} and R^{100} were harvested from pure vetch and pure rye stands and any weeds removed, while vetch and rye for the $R^{50}V^{50}$ and $R^{75}V^{25}$ treatments were separated from appropriate mixed stands and reconstituted as the mixtures noted above. Residues were dried at room temperature before reconstitution and incubation.

Cover crops applied in the incubation were analyzed for total C, N and biochemical fractions. These residue quality parameters were expected to correlate with the amount and timing of release of N from residues in incubations and field plots. A subsample of each of the residues and residue mixtures was dried at 55°C and ground to pass through a 2-mm sieve. Total C and N in residue were measured by dry combustion with a TruSpec CN Carbon/Nitrogen Determinator (Leco, St. Joseph, MI). Biochemical fractions of the residues were determined by using the neutral detergent and acid detergent stepwise fiber fractionation procedure developed by Van Soest et al.[35] using an ANKOM-automated system with filter bags (ANKOM Technology Corp., Fairport, NY). Neutral detergent fiber (NDF) includes hemicellulose, cellulose and lignin as the primary components; acid detergent fiber (ADF) consists of cellulose and lignin; and acid detergent lignin (ADL) is the component remaining after cellulose has been removed[36].

The method for the laboratory incubation was similar to that of Fortuna et al.[37]. Soil was collected from four plot replicates of the untreated control in the field experiment to a depth of 15 cm. Soil cores from each individual field replicate were composited, passed through a 2-mm sieve, and pre-incubated at room temperature for

7 days. Incubation vessels were 120-ml specimen vials (diameter 53 mm, height 60 mm). Pre-incubated soils (50 g oven dry equivalent) were weighed into the specimen vials and packed to a bulk density of 1.2 g cm^{-3}. Air-dried residues were cut into 5–7 cm lengths and were mixed into the soil with a spatula such that less than 10% was visible from the surface, to simulate rotary spading, cultivation that uses rotary motion to turn spade-shaped blades to invert the soil surface. Incubations were maintained at 40% water-filled pore space (WFPS), 18% gravimetric water content. Once a week, soil samples were weighed and less than a gram of distilled water was added to the soil surface of each vessel with a pipette to maintain WFPS at 40%. The caps were not tightly sealed, which allowed gas exchange, but slowed drying. Temperature was maintained at 25°C. There were five treatments, each with three replicates and eight sample dates, for a total of 120 incubation vessels. Specimen vials were removed and sampled 0, 7, 15, 28, 50, 70, 106 and 157 days after set-up and inorganic N, ($NH_4^+ + NO_3^-$)–N was measured. Inorganic N was extracted from 10 g soil samples with 100 ml of 1 M KCl. Samples were shaken on a reciprocal shaker for 1 h and then filtered through No. 42 Whatman filters. Aliquots were run on an automated, continuous flow QuikChem 8000 Injection Flow Analysis System (Hach Instruments, Loveland, CO). Nitrate–N was determined using the cadmium reduction method[38]. Ammonium–N was determined using a salicylate–nitroprusside method[39]. In addition, the electrical conductivity (EC) and pH of incubated soils at time intervals 7, 15, 28, 50, 70 and 157 were measured in 1:5 soil:H_2O mixtures[38].

Field study

Experimental design.

Site description. The field study was conducted from September 2008 to September 2010 on a site located (longitude 122°18′W, latitude 47°0′N) at the Washington State University Puyallup Research and Extension Center, in the Puget Sound region of western Washington, USA. The soil developed in recent alluvium and is classified as a Puyallup fine sandy loam (coarse-loamy over sandy or sandy-skeletal, isotic over mixed, mesic Vitrandic Haploxerolls). The climate in Puyallup is temperate, characterized by mild, dry summers and cool, wet winters, with a mean annual temperature of 10.9°C (1995–2010). Average annual precipitation is 1029 mm yr^{-1} (1995–2010) with approximately 75% of the precipitation received between October and March. The plot area has not been certified organic, but has been under organic management since 2004. The entire field received compost at approximately 34 Mg ha^{-1} dry matter in July 2008.

The field experiment was designed to evaluate the N contribution of a rye–hairy vetch mixture to organically grown sweet corn. The study was arranged in a randomized complete block split-plot design. Each of the four blocks contained randomized replicates of each treatment. Main plot treatments were cover crop planting date (early, late or no cover). Split-plot treatments, also randomized, were the N-fertilizer rates applied to sweet corn (0, 56, 112 and 168 kg N ha^{-1} as feather meal available N). Main plots measured 24.4 × 6.1 m, split plots measured 6.1 × 6.1 m. Early and late planting dates were mid September and early October each year and the rye–hairy vetch treatments were a 50:50 mixture by seed weight ratio. The N source was feather meal, a relatively fast release, commercially available organic fertilizer containing 12% N. Approximately 75% of the N is available in a growing season under maritime Pacific Northwest conditions[40].

Baseline soil data. Soil samples taken to a 30 cm depth prior to cover crop planting in August 2008 at the start of the field experiment, indicated that phosphorus, potassium and pH levels were uniform across the field site. Bray-1 P content was 280 mg kg^{-1} [41] and NH_4OAc extractable potassium analyzed using a Thermo IRIS Advantage HX Inductively Coupled Plasma (ICP) Radial Spectrometer (Thermo Instrument Systems, Inc., Waltham, MA) was 206 mg kg^{-1} [38]. Soil pH at the field site was 5.9[38].

Management. Common rye and hairy vetch were drilled together in the fall using a 3.1 m John Deere grain drill and chopped in the spring using a flail mower set 5 cm above the soil surface. Cover crops were incorporated using a rotary spader, a cultivator that uses rotary motion to turn spade-shaped blades to invert the top 25 cm of the soil. Seeding rate for the rye–hairy vetch mixture was 112 kg ha^{-1}; all vetch seed was inoculated with *Rhizobium leguminosarum* biovar *viceae* to ensure N fixation. Feather meal was hand-broadcast across each split plot at the appropriate rate prior to planting sweet corn. 'Golden Jubilee' sweet corn was seeded in June with a Cole Planter on 1.5 m beds with 0.76 m between-row and 0.1–0.15 m in-row spacing at the rate of 86,100 seeds ha^{-1}. Each plot was 8 rows wide. Weeds were controlled as necessary via cultivation (basket-weeder, tine-weeder and sweeps) and hilling. Irrigation was supplied as needed via sprinklers on 2.4 m risers. Approximately 20–25 mm of water per irrigation was applied on four dates in 2009 and on two dates in 2010. After harvest, sweet corn stover was flail mowed and incorporated with a rotary spader, and the plots were prepared for cover crops for the next season. Dates of field activities and sampling are listed in Table 3.

Field measurements

In late April cover crop above-ground biomass was measured by harvesting a 6.1 × 0.91 m swath approximately 5 cm above the soil surface with a flail mower. The harvested forage from each plot was weighed wet, and a subsample (approximately 500 g) was collected and oven dried at 55°C to determine dry matter. The remaining

Table 3. Dates of sampling and field activities.

	2008–09	2009–10
Early planting	September 17	September 14
Late planting	October 2	September 29
Cover crop biomass sampling	April 27	April 26
Field chopped	May 5	May 10
Residue incorporated	May 7	May 10
Soil sampled for NO_3–N	June 2	June 1
Fertilized	June 2	June 7
Corn planted	June 4	June 16
Soil sampled for NO_3–N	June 26	July 18
Ear leaf sampled	August 4	September 7
Corn harvest	August 27	September 27
Soil sampled for NO_3–N	September 9	September 28
Residue incorporated	September 10	September 30

residue was returned to the plots. The dried subsample was ground to pass a 2-mm sieve and analyzed for total C and N using a TruSpec CN Carbon/Nitrogen Determinator (Leco). Three additional subsamples were hand clipped from representative 0.25 m^2 quadrats in each plot, and proportions of rye, vetch and weeds were determined[42].

Soils were sampled for NO_3^-–N analysis on three occasions: 3–4 weeks after cover crop incorporation, 3–4 weeks after corn planting, and after harvest (Table 3). Monitoring soil NO_3^- enabled us to determine the lag between cover crop incorporation and N mineralization, as well as the potential N credit provided by the cover crop mixtures. Ten soil samples (0–30 cm depth) were collected randomly from each plot with a 2.5-cm diameter probe and composited at each sampling date. Soil was immediately dried at 30°C to stop microbial activity. Soil moisture was determined on a subsample dried at 105°C. Dried soil was passed through a 2-mm diameter screen opening and (NO_3^-)–N content determined. Nitrate–N was extracted from 10 g soil samples with 100 ml of 1 M KCl. Samples were shaken on a reciprocal shaker for 1 h and then filtered through No. 42 Whatman filters. Aliquots were run on an automated, continuous flow QuikChem 8000 Injection Flow Analysis System (Hach Instruments). Nitrate–N was determined using the cadmium reduction method[38].

The in-season N status of corn ear leaves in the zero-feather meal plots was measured at silking. Ear leaves from 10 plants from the middle four rows of each plot were removed, dried at 55°C, ground to pass through 2-mm screen opening, and total C and N determined via dry combustion with a TruSpec CN Carbon/Nitrogen Determinator (Leco).

At harvest, ears of corn were hand picked from within a 6.1-m section of an inner row in each plot. Fresh weight and number of corn ears were recorded. Total above-ground biomass of sweet corn plants was removed from a 6.1-m section of a second inner row in each plot and the fresh weights recorded. Ten plants randomly chosen from each biomass sample were run through a wood chipper in the field, mixed, subsampled (approximately 900 g wet weight), dried at 55°C, and ground to pass through a 2-mm screen opening for total C and N analysis. Water content was determined and used to calculate dry weight. All plant residues (cover crop, sweet corn ear leaves and sweet corn biomass) were analyzed for total C and N by dry combustion with a TruSpec CN Carbon/Nitrogen Determinator (Leco).

Data analysis

Net N released from cover crop residues during the laboratory incubation was calculated by subtracting inorganic N content of the control soil from total inorganic N content of each cover crop treatment. The percent N release was calculated by dividing net N released from cover crop residue by total N content of residue applied. Net N mineralization data were fitted to a simple exponential model (Eqn 1)[22], which assumes a single pool of potentially mineralizable N, mineralized at a rate proportional to its concentration:

$$N_m = N_o[1 - \exp(kt)] \quad (1)$$

where N_m is mineralized N at time t, t is the incubation time, N_o is the potentially mineralizable N pool, and k is the mineralization rate constant. Curve fitting of N mineralized was approximated by least-squares iteration with the SAS NLIN procedure (SAS 9.2, SAS Institute, Cary, NC, 2008). The confidence intervals for each parameter were used to test for significant differences among parameters for each treatment. Non-overlapping confidence intervals indicate significant differences in parameters. A two-factor factorial analysis of variance was performed using SAS PROC MIXED to determine the effect of the ratio of rye to hairy vetch biomass on N mineralization using SAS 9.2 (SAS Institute, 2008). Two- and three-way analyses of variance (ANOVAs) were performed on cover crop biomass, the percent of rye and hairy vetch in the total cover crop biomass, total residue N content, and soil nitrate–N, using the SAS PROC MIXED procedure. Means were considered significantly different at the Bonferroni adjusted P-value of 0.05. Pearson correlation was used to test for monotonic trends between: net N mineralization and residue quality, EC and soil nitrate–N, and EC and pH in the SAS CORR procedure. Growing degree days were calculated per the Baskerville–Emin method[43], using base temperature of 4°C.

Results and Discussion

Nitrogen release from cover crop residues in the laboratory incubation

Contrary to Kuo and Sainju[13], we did not observe net N immobilization at any time during the incubation or field

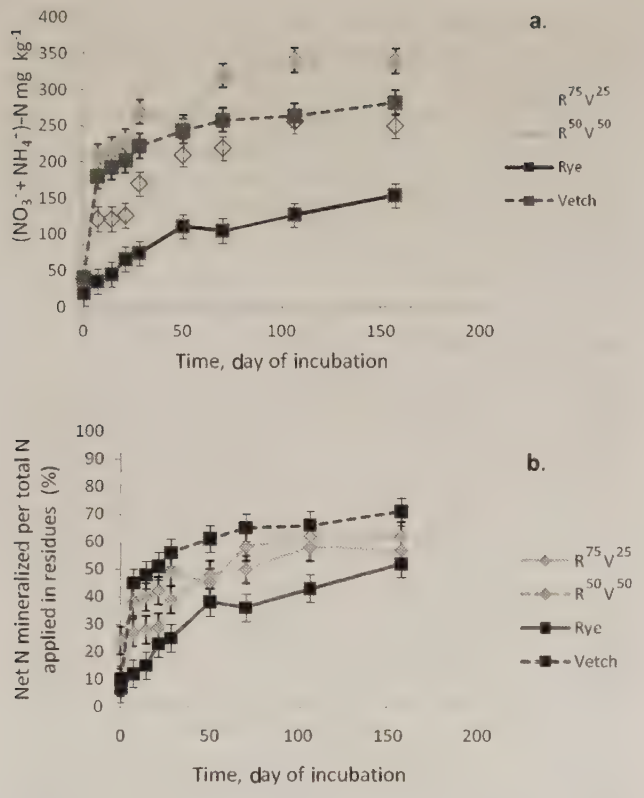

Figure 1. Net nitrogen (N) mineralized from rye (R) and hairy vetch (V) residues as mg N kg^{-1} soil (a) and as percentage of residue N added (b) $P \leq 0.05$.

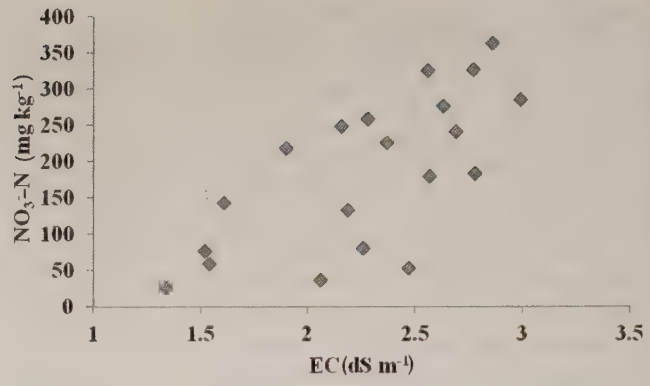

Figure 2. The relationship between EC (dS m^{-1}) and nitrate–N (NO_3^-–N) in soil incubated for 150 days with residue amendments that included: rye, hairy vetch and rye–hairy vetch blends (50% vetch:50% rye and 75% rye:25% vetch). Pearson correlation was used to test for monotonic trends. The Pearson correlation coefficient (0.70) positively correlated EC and NO_3^-–N at $P < 0.001$.

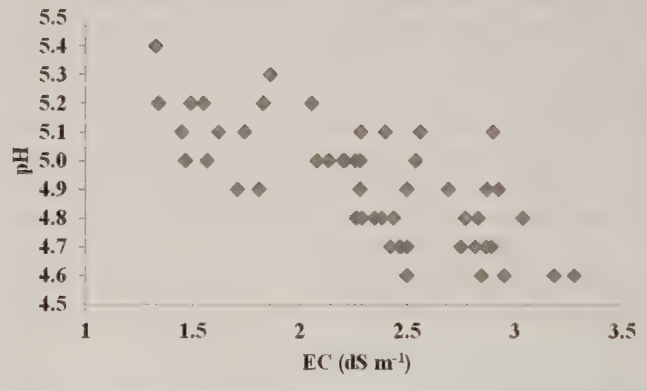

Figure 3. The relationship between EC (dS m^{-1}) and pH in soil incubated for 150 days with residue amendments that included: rye, hairy vetch and rye–hairy vetch blends (50% vetch:50% rye and 75% rye:25% vetch). Pearson correlation was used to test for monotonic trends. The Pearson correlation coefficient (−0.73) negatively correlated EC and pH at $P < 0.001$.

experiments. In our incubation study, the N concentration for R^{100} (the treatment with the highest C:N ratio) (Table 2) was above the critical value for net N mineralization observed by Odhiambo and Bomke[44] (14 g kg^{-1}), but lower than that reported by Kuo et al.[45] (18 g kg^{-1}). The literature demonstrates that there is a wide variation in the critical N concentration necessary for net N mineralization, ranging from 10 to 18 g kg^{-1}.[45,46] The net accumulation of $(NH_4^+ + NO_3^-)$–N during the N mineralization incubation study was positive in all cover crop treatments at all sample dates (Fig. 1a), indicating that all the residues had C:N ratios sufficiently narrow (13–26) to allow for mineralization of N (Table 2).

Nitrogen accumulation curves (Fig. 1a) show a rapid mineralization phase for all cover crop treatments during the first 28 d followed by a plateau. This plateau is partly due to the salt effect that occurs as soil NO_3^-–N builds up, resulting in an increase in EC (Fig. 2) and a decrease in pH (Fig. 3) associated with an increase in H^+ ion activity produced during the process of nitrification. Residue amendments that contained vetch or a mixture of rye: vetch blends reduced the pH of the soil from a baseline of 5.5 in unamended control treatments by approximately half a pH unit. The EC of untreated incubated soils ranged from 0.50 to 0.73 dS m^{-1}. Residue amended soils had EC values ranging from 1.3 to 3.0 dS m^{-1} after day 7 of the incubation. Although these values are considered only slight to moderately saline, EC values within this range decrease mineralization rates (C and N), nitrification and yields in most agricultural and horticulture crops (Smith and Doran)[47]. The negative relationship between pH and EC values in amended soils (−0.72, $P = 0.001$) and the positive correlation (0.70, $P = 0.001$) between EC and nitrate–N during the duration of our incubation measured via a Pearson correlation have been observed by other researchers across a range of field and laboratory conditions[47–49]. Application of single and repeated injections of sludge increased EC and decreased pH in a terraced cropland of eastern Nebraska, USA[49]. During the study, a single sludge application temporarily reduced the soil pH by 0.40 unit. Amending and incubating soils with vetch and rye:vetch blends in this study resulted in a

Table 4. First-order parameters for net N mineralization[1] from cover crop residues (R = rye, V = vetch) in a laboratory incubation study.

Cover crop treatment	N_o[2] (Mg kg^{-1})	k (day^{-1})	MRT (days)
V^{100}	209 b[1]	0.091 a	11
$R^{50}V^{50}$	248 a	0.061 b	16
$R^{75}V^{25}$	200 b	0.029 c	34
R^{100}	135 c	0.018 c	56
Control	93 d	0.006 d	167

[1] A non-linear regression equation was fitted. An ANOVA analysis was run on the individual parameters N_o and k. Values within columns followed by the same letter are not significantly different at $P \leq 0.05$.
[2] N_o, potentially mineralizable N pool; k, decay constant; MRT, mean residence time.

similar drop in pH. Increases in EC under laboratory conditions result in more dramatic decreases in microbially driven processes such as mineralization, nitrification and denitrification during laboratory incubations[48]. The relative effect of pH versus EC on microbial processes in this study was confounded by a low baseline soil pH and ions other than nitrate (Cl^-, SO_4^{2-}, CO_3^{2-}, PO_4^{3-} and OH^-), which contributed to increases in EC. Future studies will include measurements of gaseous N loss, and treatments that provide control pH and EC concentrations.

The proportion of N mineralized increased as the proportion of vetch in the mixtures increased (Fig. 1b), as expected based on the C:N ratios of the mixtures (Table 2). As little as 25% vetch residue combined with rye showed a significant increase in mineralized N compared with rye alone (Fig. 1a). This result is consistent with other findings[13]. The amount of N mineralized in the laboratory incubation depended on both the proportion of cover crop N mineralized, and the amount of cover crop N added to the incubation vessels. Because the incubations were set up at rates proportional to field yields of the various mixtures from Lawson[16], the rye:vetch mixtures ($R^{75}V^{25}$ and $R^{50}V^{50}$) were amended at higher residue rates than the V^{100} treatment (Table 2). As a result, total N mineralized from the pure vetch and the $R^{75}V^{25}$ treatment was similar, even though the proportion of N mineralized from the pure vetch was greater (Fig. 1a and b).

Net N mineralization from cover crop residues fit a single-pool first-order kinetic equation. The size of the mineralizable pool of N (N_o) for $R^{50}V^{50}$ (248 mg kg^{-1}) was nearly double R^{100} (135 mg kg^{-1}) (Table 4). The other treatments ($R^{75}V^{25}$ and V^{100}) had similar N_o values that were slightly smaller than $R^{50}V^{50}$. N_o was influenced by the proportion of the rye and vetch residues and the amount of residue applied. The rate constant k was highest for V^{100} and decreased as the rye component in the mixtures increased. Although N_o was similar between V^{100} (209 mg kg^{-1}) and $R^{75}V^{25}$ (200 mg kg^{-1}), the rate constant k was significantly smaller for $R^{75}V^{25}$, 0.029 versus 0.091 day^{-1} (Table 4). This suggests that although similar total N release is expected for these treatments, the timing of release is much more rapid for V^{100}. The slower release of N from $R^{75}V^{25}$ residues may be better timed with the critical period of crop N uptake and therefore enhance N uptake efficiency.

Effect of residue C:N and fiber fractions on N release

Residue C:N ratio was negatively correlated with net N release from rye, vetch and mixtures of rye–hairy vetch cover crops for the first 70 days of the 150-day incubation (Pearson correlation coefficients ranged from −0.83 to −0.93), which is consistent with other studies[46,50]. The C:N ratio of residues was a better predictor than fiber characteristics. In this study, NDF was negatively, but more weakly correlated to net N mineralization through the first 70 days of the incubation. Pearson correlation coefficients between NDF and N mineralization ranged from −0.64 and −0.83. Others have reported that the NDF fraction and hemicellulose, a component of the NDF fraction, are sometimes weakly related with N release from plant residues, and neither are particularly good predictors of N release[33,50]. Our data indicate that N release patterns from cover crop residues with narrow C:N ratios can be predicted based on the C:N ratio and correlated using growing degree days (Table 5). Other researchers have used similar predictors[51].

Field study: cover crop biomass, plant composition and quality

Planting date and year had a significant effect on cover crop biomass accumulation, species composition, and C:N ratio (Table 6). Rye was killed at the booting stage and vetch in the vegetative stage, which reduced the potential for high C:N ratios. The number of growing degree days accumulated between planting and harvest dates was greater in 2010 and may account for the difference in biomass production between years (Table 6). The ratio for the early planted cover crop biomass in 2009 was approximately 3:1 rye–hairy vetch, which is consistent with cover crop stands grown in a similar study at WSU Puyallup, WA, USA[16]. Delaying the planting date decreased the amount of hairy vetch in the biomass relative to rye in 2009. The early-planted cover crops in our study had biomass yields similar to other studies in cool climates, but total N was near the low end of the range, clustered with other data from western Washington (Table 1). Cool conditions in western Washington during the establishment and spring growth phases may limit hairy vetch production, resulting in lower N uptake. In 2010, cover crop stands were dominated by rye, which accounted for 95% of the total biomass. It is unclear why hairy vetch did not grow well in 2010. Lower tissue N concentrations and higher C:N ratios in 2010 compared to

Table 5. Correlation of growing degree days elapsed between cover crop incorporation and soil sampling in incubation and soil and ear leaf sampling in field.

	Laboratory incubation		Field trial		
Year	Days	Degree days	Sample	Date	Degree days[a]
2009	14	288	Pre-plant corn, soil	June 2, 2009	249
	28	576	Soil and ear leaf	June 26, 2009	546
	70	1439	Post-harvest soil	September 9, 2009	1579
2010	7	144	Pre-plant corn, soil	June 1, 2010	186
	28	576	Soil and ear leaf	July 18, 2010	727
	70	1439	Post-harvest soil	September 28, 2010	1661

Table 6. Cover crop (R = rye, V = vetch) biomass yield, total N uptake, tissue N concentration, C:N ratio, and dry matter composition for residues sampled from the field in 2009 and 2010.

						Dry matter composition		
Study	Cover crop treatment	Biomass ($Mg\,ha^{-1}$)	N uptake ($kg\,ha^{-1}$)	N concentration ($g\,kg^{-1}$)	C:N Ratio	Rye (%)	Vetch (%)	Degree days
Field–2009	Early[1]	2.76 a[2]	62 a	22.3 a	19 a	78 b	22 a	843[3]
	Late	1.50 b	34 b	22.7 a	18 a	87 a	13 b	699
Field–2010	Early	4.81 a	76 a	15.9 a	27 a	95 a	5 a	1023
	Late	2.90 b	50 b	17.5 b	25 b	95 a	5 a	856

[1] Early and late cover crops were seeded at a 50:50 rye–hairy vetch proportion in the field.
[2] An ANOVA was run in SAS Proc Mixed. Values within a column and year followed by the same letter are not significantly different at $P \leq 0.05$.
[3] Degree days elapsed between cover crop planting date and incorporation. Calculated on a base temperature of 4.4°C.

Table 7. The effect of cover crop and feather meal rate on soil nitrate (NO_3^-)–N concentration measured in early June, at pre-sidedress nitrate test (PSNT) time, and post-harvest. Zero-feather meal plots only were sampled in early June and at PSNT.

			Feather meal available N rate ($kg\,ha^{-1}$)					
Year[1]	Significance	Cover crop treatment	0 Early June	0 PSNT	0	56	112	168
						Post-harvest		
			Soil (NO_3^-)-N, $mg\,kg^{-1}$					
2009	*	Early	8 a	18 a[1]	3 d	5 c	6 bc	23 a
		Late	7 a	16 b	2 d	3 c	7 bc	8 bc
		None	7 a	15 b	2 d	3 c	4 c	10 b
2010	NS	All[2]	6	15	4 c	5 c	8 b	13 a

[1] ANOVA was run in SAS PROC MIXED. Values followed by the same letter within each test date and year are not significantly different at $P \leq 0.05$. *Indicates significance at $P \leq 0.05$. NS = not significant.
[2] There were no significant differences among treatments in 2010.

2009 can be accounted for by the low amount of hairy vetch biomass (5%) present in those stands (Table 6). The cover crops effectively competed with weeds both years, with weeds averaging 3% of total biomass in 2009 and less than 1% in 2010.

In-season soil nitrate and sweet corn ear leaf N

Cover crops were not a major source of soil nitrate for sweet corn uptake. The early-planted cover crop had a significant, but small effect on field soil NO_3^- levels in the zero-feather meal plots at the time of the pre-sidedress nitrate test (PSNT) in 2009, and no effect in 2010 (Table 7). The lack of response to cover crop derived N in 2010 was likely due to the lower proportion of hairy vetch in the cover crop mixtures compared with 2009. Nitrate concentrations in the late-planted cover crop treatment were not different from the control in either 2009 or 2010. In both years, soil NO_3^- in the zero-feather meal plots was highest at the time of the PSNT test

Table 8. Sweet corn fresh ear yield and above-ground dry matter as affected by feather meal for 2009 and 2010 field seasons averaged over all cover crop treatments.

Year	Feather meal available N ($kg\,ha^{-1}$)	Ear fresh weight ($Mg\,ha^{-1}$)	Above-ground dry matter ($Mg\,ha^{-1}$)	N uptake ($kg\,ha^{-1}$)
2009	0	18 c[1]	7.79 b	110 d
	56	21 b	8.28 b	143 c
	112	22 ab	8.54 ab	166 b
	168	24 a	9.04 a	189 a
2010	0	12 b	6.76 c	90 d
	56	16 a	7.54 b	119 c
	112	18 a	7.85 b	134 b
	168	17 a	8.41 a	150 a

[1] ANOVA was run in SAS PROC MIXED. Values followed by the same letter within a year and column are not significantly different from each other at $P \leq 0.05$.

(approximately 8 weeks after cover crop incorporation in 2009 and 11 weeks in 2010), and lowest immediately after sweet corn harvest (Table 7). All of the nitrate-N levels measured in the PSNT test were less than the sufficiency level for silage corn ($25\,mg\,kg^{-1}$) developed for manured systems in the maritime Pacific Northwest, USA[52].

Ear leaf nitrogen is another in-season test for N sufficiency (Table 8). Regional standards for the Pacific Northwest report that sweet corn is N deficient when ear leaf N concentrations are below $28\,g\,kg^{-1}$ [53]. The early cover crop treatment (without feather meal) had significantly higher ear leaf N concentrations (28.6 and $24.7\,g\,kg^{-1}$ in 2009 and 2010) at silking compared with the no cover crop treatment without feather meal (26.0 and $21.2\,g\,kg^{-1}$ in 2009 and 2010). Sweet corn in 2009 on the early cover crop treatment was the only treatment that contained sufficient N based on ear leaf analysis.

Sweet corn N uptake, dry matter production and fresh ear yield

Cover crop N contribution to the sweet corn was relatively small compared with soil N and feather meal N. Cover crop effects on sweet corn above-ground biomass, ear yield and N uptake were small and not statistically significant (data not shown; $P = 0.05$). Total N accumulated in the above-ground biomass of the early (mid September) planted cover crop was 70–75 $kg\,ha^{-1}$ with ANR in the sweet corn estimated as 15–30 $kg\,ha^{-1}$. Planting in October reduced cover crop N accumulation to 40–50 $kg\,ha^{-1}$ and ANR to less than 10–20 $kg\,ha^{-1}$. Feather meal effects were significant ($P < 0.05$) for all three measurements (Table 8), with an ear yield plateau occurring at the 112 $kg\,ha^{-1}$ rate. The only N effect of cover crop was apparent in the post-harvest soil nitrate samples for the early cover crop in 2009 (Table 7), and then only at the highest feather meal rate, which was in excess of the peak rate for ear yield. There were no significant interactions between cover crop and feather meal treatments.

The lack of significant response to cover crops was likely a combination of low N supply from the cover crops (Table 6) and high background available N in the soil. Soil background N was 87–98 $kg\,N\,ha^{-1}$ as measured by N uptake into sweet corn in the unfertilized, no-cover crop control plots. These measurements are consistent with field data from previous experiments with similar soil and environmental conditions[40,54].

Nitrogen supply from cover crops—comparison of field and laboratory incubations

The incubations showed that the composition of the cover crop mixtures had a large effect on the initial mineralization rate, with pure hairy vetch (C:N = 13) releasing N at three times the rate of the $R^{75}V^{25}$ mixture (C:N = 21). The slower rate of N release from the rye–vetch mixtures may result in N release that is better timed with crop N uptake, increasing N uptake efficiency. In the field, cover crop N contribution to sweet corn was small compared with soil and feather meal N because of low cover crop N accumulation and the high proportion of rye in cover crop mixtures.

The number of growing degree days between cover crop incorporation and sweet corn harvest varied between year (1430 days in 2009 and 1650 days in 2010), but was comparable to the number of growing degree days accumulated over the first 70 days of the laboratory incubation (1439 days). Combining sweet corn N uptake data from the field trial with incubation N release data provided an estimate of cover crop available N. In 2009, early cover crop N content was 62 $kg\,ha^{-1}$ and the composition was similar to a mineralization rate of 50% over a 70-day period that was measured in the $R^{75}V^{25}$ treatment of the incubation. This treatment provided an estimated 31 $kg\,ha^{-1}$ of available cover crop N. In 2010, N content was 76 $kg\,ha^{-1}$ with a composition similar to pure rye (70-day mineralization rate of 36%), providing an estimated 28 $kg\,ha^{-1}$ available N. The late-planted cover crops provided less than 20 $kg\,ha^{-1}$ of available N using similar calculations. Therefore, the N contribution of cover crops was insufficient relative to the seasonal N requirement of sweet corn.

Conclusions

The rye–hairy vetch blends grown in this study had modest biomass yields, similar to other regions with cool spring and fall climates, but contained less hairy vetch and had lower N accumulation than reported from other parts of the USA. Relatively cool temperatures during the critical times of establishment in the fall and biomass

production in April likely favored rye over the hairy vetch. We did not observe immobilization of N in the pure rye incubation or in field plots containing rye and rye–hairy vetch mixtures, either in the field or in the laboratory incubations. This may be because the rye did not mature beyond the boot stage at the time of termination. The incubations showed that the composition of the cover crop blends had an effect on the initial mineralization rate, with pure hairy vetch releasing N at three times the rate of a 75% rye–25% hairy vetch ($R^{75}V^{25}$) blends. The slower rate of N release from the rye–vetch mixtures may result in N release that is better timed with crop N uptake, increasing N uptake efficiency. Our study indicates that there is the potential to manage rye, or predominantly rye, cover crop mixtures for improved soil quality without creating conditions that result in N immobilization in the field during the period of crop N uptake. While cereal–legume cover crop mixtures have been shown to be a reliable source of N to cash crops in other parts of the USA, this does not appear to be the case in the maritime region of the Pacific Northwest, USA. Long-term additions of cover crop biomass contribute to soil organic N, but cover crops were an insufficient source of seasonal plant available N in our system. Future research should address short-term increases in EC that influence the rates of microbial processes controlling mineralization and gaseous losses during incorporation of cover crop amendments under field and laboratory conditions.

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Organic label as an identifier of environmentally related quality: A consumer choice experiment on beef in Italy

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Abstract

This paper uses a hypothetical choice experiment to investigate Italian consumer preferences and willingness-to-pay (WTP) for organic, conventional and genetically modified (GM)-fed beef, utilizing intrinsic, search cues (price, color and visible fat) and extrinsic, credence cues. Data are gathered from three different locations in northern, central and southern Italy using a sequential Bayesian approach. Results showed that consumers attach higher value to organic meat. WTP for GM-fed beef, which is not yet sold in Italy, is well below current conventional beef prices. Organic beef is attractive to consumers because it is associated with higher animal welfare standards and environment-related issues (food miles and biodiversity preservation). No differences are found in marginal WTP estimates by gender, age, education, being a parent or having a higher level of knowledge about organic production. Ethical/environmental issues (credence cues) appear to be more relevant in explaining variation in WTP for organic beef than ordinary product characteristics (search cues).

Key words: consumer choice experiment, willingness-to-pay, animal welfare, country of origin, local breeds, ethical and environmental values, organic beef, marketing

Introduction

The demand for organic meat products is growing in all of Europe in response to an increasing concern for safety, human health and environmental issues[1]. Past studies[2,3] have shown that consumers assign to organic products a high level of food quality and safety, as well as other benefits. For instance, organic beef is associated with lower 'food miles' than conventional beef. Consumers tend to associate organic beef production with good health, animal welfare and concern for the environment[4]. Studies carried out in different European countries have found that consumers' motivations in the purchase of organic food is mainly related to concerns for health, environment and animal welfare[5–11].

According to the quality guidance approach[12], consumers need information on the quality characteristics of the product in order to evaluate food quality. They receive this information through cues, which are pieces of information used to form quality expectations[13]. Physically related aspects (color, visible fat, etc.) are named intrinsic cues, whereas product-related aspects that are not physically part of it are extrinsic cues (breed origin, animal welfare, etc.).

Researchers have separated those attributes which are important in explaining demand for organic food products in two quality attributes groups: credence and experience[3,14]. The latter are attributes that consumers can only determine during or after consumption, but not at the time of purchase, such as the tenderness of a steak. Credence attributes, instead, cannot be discerned by consumers even after consuming the product, such as animal welfare or (lower) environmental impact. According to the multi-attribute approach[15] consumers combine experience and credence cues into a composite perception of overall quality. Thus, consumers who prefer organic to conventional food are likely to perceive and highly value credence attributes of organic products and consequently are expected to be willing to pay a higher price for these characteristics.

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In our study we considered four credence attributes related to beef choices: (1) the organic mode of production, contrasted to conventional and genetically modified (GM)-fed beef; (2) production methods respectful of animal welfare as opposed to conventional methods; (3) country of origin, which is also a proxy for food mileage; and (4) breed origin, as a proxy for biodiversity conservation.

Of course, these variables are not exhaustive of the broad range of environmental and ethical attributes that determine food (and beef) choice by consumers. They have been selected among those mentioned by Padel and Gössinger[16] because they could be easily related to beef choice by consumers.

The organic attribute—*per se*—is related to both quality and safety concerns, and embeds many environmental and ethical-related attributes. In this study, we contrasted beef produced organically with both conventional and GM-fed beef. Conventional animal farming has been seriously challenged by recurrent crises (such as mad cow disease, foot and mouth disease, scrapie, etc.) which resulted in increased concern by consumers for food safety.

Previous studies have shown that diffidence toward beef from GM-fed cattle is particularly high in Europe[17,18]. In 2007, the EC issued a revised EC Regulation for organic food and farming no. 834/2007, which came into force on January 1, 2009. The Regulation contains many novelties —generally welcomed by the organic sector—but also introduces a very controversial reference to mandatory labeling of GM food. The exclusion of GM ingredients or feed in organic production is bound to be below the same threshold applied to conventional farming, i.e. not more than 0.9%. At the same time, the reference to the GM labeling scheme is thought to 'provide for numerous loopholes which allow for the presence of GM materials beyond 0.9 percent in organic products'[19], since unwanted components such as those introduced accidentally, are believed not to trigger mandatory GM labeling. Therefore, the risk of contamination of organic food by GM materials is a serious threat, and the coexistence of organic and GM crops is a concern for both organic farmers and consumers.

Animal welfare is widely recognized as one of the most important credence characteristics for consumers. In their study on preferences for pork product and process attributes, Caracciolo et al.[20] found that generally European consumers take account of animal welfare attribute more than other intrinsic product characteristics. Nevertheless, this concern seems to vary across European countries: northern European consumers have higher propensity toward sustainable pig farms, while consumers from southern Europe show greater attention to meat fat quality. Moreover, Cicia and Colantuoni[21] in their meta-analysis for willingness-to-pay (WTP) on meat traceability, indicate Animal Welfare with Food Safety and Country of Origin as the most requested attributes. In our study, for simplicity, we measured animal welfare using a cue that any consumer could easily understand: were the cattle allowed to range freely (in pastures or paddock) or were they confined and chained? In Italy, most beef cattle are not free-range, and are intensively raised in confined stables.

Country of origin is a quality attribute related to the concept of traceability, an aspect of increasing importance in consumer's choice and behind the motivation to purchase[22]. In a previous study, beef origin was identified as the most important attribute, followed by information about animal welfare and price, in purchase decisions[23]. A more recent study[24] shows that beef of national origin, which is produced with protocols respectful of animal welfare and production processes ensuring product quality, is perceived as a highly attractive product by the majority of the consumers. We note that the WTP for country of origin may largely differ between countries[21] and even within each single country, depending on cultural differences and on other macroeconomic variables (i.e., inflation and per-capita income). In terms of observable characteristics of beef steak, our study includes color, visible fat and price, which previous studies identified as the most significant intrinsic quality characteristics for beef[2,25].

Against this background, this paper reports some preliminary results of a case study on consumer's preference for organic beef production by means of non-soy-based feed. An unlabeled choice experiment was conducted to investigate consumer preferences and WTP for organic, conventional and GM-fed beef with respect to the intrinsic and extrinsic cues mentioned before. From these choice data we investigate the role of the organic label as an identifier of environmentally related beef quality in three Italian locations. As preference measurements are quantitatively expressed in terms of trade-off between one attribute versus another, we focus on a trade-off that would be meaningful to most, which is that with money, so that we obtain estimates of marginal WTP.

Data and Methodology

This section reports on the choice experiment used for data collection and on the statistical analysis of the data collected in order to retrieve structural estimates of utility functions. Choice experiments are used in applied economics to derive structural estimation of consumer preferences for products currently not available in the market, such as in our case GM beef. In Italy, in 2008, this production method was only hypothetical because livestock could not be fed GM-labeled feed (i.e., feed containing more than 0.9% of GM materials). Choice in the experiment takes the form of selecting the favorite alternative out of a set of three. The interpretation of observed choices is based on the Lancasterian theory of

Table 1. Choice experiment attributes and levels.

Attributes	Levels
Color	Pink
	Red
Production method	OGM
	Conventional
	Organic
Animal welfare	Box
	Open
Place of production	Italy
	Abroad
Price (€/kg)	12
	24
	36
Visible fat	No
	Yes
Breed origin	Local
	No local

value[26], which purports the total value of a product as the sum of the values of its valuable characteristics. By observing people's choices in a number of experimentally designed choice contexts, statistical models of choice are estimated and interpreted using random utility theory[27].

Data collection and sample

Experiments were conducted in three different Italian locations in northern (Udine), central (Ancona) and southern Italy (Potenza), during February–March 2008. Apart from their geographical location, we chose these cities because the 'average' Italian is more likely to live in a town with a population between 50,000 and 200,000 (such as those we have surveyed) than in large cities such as Rome, Milan or Naples with a population over a million. Italy has only three cities with a population larger than 1 million, but has a population of 60 million. Even if one considers the larger city conurbations, fewer than 20 million people live around large urban centers. Medium-sized towns were chosen because the majority of Italians live in urban centers with these characteristics.

The preference structure behind the attributes of beef was unknown *a priori*. However, the set of intrinsic and extrinsic cues embodied in the attributes selected to describe the beef steaks were either determined by the objective of the study or by the importance that they have in determining beef choice. Seven attributes were selected to describe steaks: color, visible fat, production method, animal welfare, place of production, breed origin and price. All selected attributes and their levels are reported in Table 1. The three price levels were determined on the basis of the average selling price for beef in the three surveyed towns: the basic price (12 euros/kg) was the average price for conventional beef, while the second level (24 euros/kg) was the average price for organic beef. The third level (36 euros/kg) was a price at which most people were expected to refuse to purchase 1 kg of steak. Color and visible fat instead were used as they are often used as indicators of quality and taste, depending on people's taste and state of health, some like more and some less visible fat in their steaks. All the other credence attributes were dictated by the objective of our research.

Since optimal experimental design for preference-based models of choice depends on the underlying preference structure in the population, the overall sample was subdivided in three sequential stages so as to gradually build up an adaptive design. Each stage in the sequence allowed for more information to be included in the previously established prior stage and was hence embedded in the experimental design of the subsequent stages. This gave the experimental design the ability to become gradually more targeted to the specific preference structure that the survey was uncovering as more information was gathered from the field. This is a form of adaptive design at the population, rather than the individual, level.

More specifically, the survey instrument and the allocation of attribute levels to alternatives were designed using a sequential Bayesian approach[28], tested in a pilot study and then refined after each stage of data collection and optimized for Bayesian WTP-efficiency[29,30]. All designs involved 36 choice tasks orthogonally blocked in three blocks of 12 each and were obtained using the Ngene software[31]. Each design was a fraction of the full factorial that was optimized on the basis of the a priori information available that far, and the assumption of a specific statistical model of reference [the multinomial logit (MNL) model] as well as the specific statistic of interest from this model (the minimization of the sum of the marginal WTPs, see[29,30]). A sample of 145 subjects (50 each in Udine and Potenza, 45 in Ancona) completed the sequence of choices, for a total of 1740 choices. The panel formulation was accounted for in the analysis using a mixed logit (MXL) model whereby the sequence of choice by each respondent shared the same preference structure and hence were correlated.

Respondents were recruited among generic food buyers. A quota sampling method was followed, with stratification criteria including age, consumption frequency of beef and organic products, in order to have enough variability between the subgroups. These were almost equally distributed among three age groups: $\leqslant 40$; $40 < \text{age} \leqslant 60$; >60. To be recruited subjects had to consume beef at least 1–2 times a month as well as occasionally (not regularly) consume organic products. Sample composition is reported in Table 2. As expected, given that the sample was not chosen to be representative; females are slightly overrepresented, while younger people are underrepresented. Besides, as often is the case with survey data, less educated people are underrepresented.

During the choice experiment, respondents were asked to choose between two types of beef steaks or neither of

Table 2. Summary statistics on characteristics of respondents.

Variables	Levels	No. of respondents	%	Population[1]%
Gender	Female	77	53.1	51.4
	Male	68	46.9	48.6
Age	$\leqslant 40$	52	35.9	76.3
	$40 <$ years $\leqslant 60$	49	33.8	15.6
	>60	44	30.3	8.1
Children	Yes	20	13.8	n.a.
	No	125	86.2	n.a.
Education	No education/primary school	12	8.3	25.0
	10 years school	18	12.4	31.6
	A level	58	40.0	32.7
	College, University	57	39.3	10.7
Knowledge	High	90	62.1	n.a.
	Low	55	37.9	n.a.

[1] Source of population data: ISTAT data warehouse (http://www.istat.it).

Table 3. Example of choice experiment task.

	Alternative 1	Alternative 2	No-buy
Color	Pink	Red	
Production method	Conventional	Organic	
Animal welfare	Box	Open	
Place of production	Italian	Abroad	
Price (€/kg)	16	32	
Visible fat	No	Yes	
Breed origin	Local	Not local	
Choice	❑	❑	❑

the two (no-purchase option). All experimental sessions were conducted in the sensory laboratory of the local university, although this was not a sensory-based experiment. Subjects who agreed to participate were asked to reach the university building at an arranged date and time. Upon arriving at the venue, respondents received a participation fee and filled in a questionnaire collecting their socio-demographic characteristics and their self-reported knowledge of organic products. Each respondent was then asked to identify their favorite alternative in each of 12 choice tasks, each including a no-buy option[32]. This option was included in order to make the choice more realistic compared to a normal purchase situation[33]. In order to reduce hypothetical biases a cheap-talk script was used[34]. Table 3 shows an example of choice task used in the survey.

Econometric methods

We explain the probability of selecting the favorite alternative from each choice set presented to the respondents using discrete choice models. So the indicator for choice is our dependent variable. Discrete choice models based on random utility theory allow researchers to derive WTP estimates for attributes describing product profiles and evaluated by respondents in a series of experimentally designed choice tasks. The behavioral foundations of this method are based on Lancastrian consumer theory[26]. According to Lancastrian consumer theory, utilities for goods can be decomposed into separable utilities for their characteristics or attributes, whereas random utility theory[27] suggests that individuals acting rationally should try to maximize their utility and hence would select the alternative that yields the highest utility. However, only the deterministic portion of such utility V_{ij} is observable by the researcher, who treats the remaining unobservable part e_{ij} as random, where i denotes the respondent and j denotes the alternative.

So, the total utility of the beef steak described in alternative j to individual i from the researcher's perspective is represented as the sum of two utility components:

$$U_{ij} = V_{ij} + e_{ij}$$

The systematic component V_{ij}, can be further approximated by a linear function of steak attributes in the vector X_{ij} and the population utility weights for each attribute collected in the vector β:

$$V_{ij} = \beta' X_{ij}.$$

The probability that the individual prefers a given beef steak j to any other options available can be expressed as

the probability that the utility associated with that steak is higher than the utility of all other k options.

In our case the indirect utilities structure for the three alternatives was:

$$V_1 = \beta_1 \text{RED} + \beta_2 \text{FAT} + \beta_3 \text{ORG} + \beta_4 \text{CONV} + \beta_5 \text{FREE} + \beta_6 \text{LOC} + \beta_7 \text{ITA} + \beta_8 \text{PRICE}$$

$$V_2 = \beta_1 \text{RED} + \beta_2 \text{FAT} + \beta_3 \text{ORG} + \beta_4 \text{CONV} + \beta_5 \text{FREE} + \beta_6 \text{LOC} + \beta_7 \text{ITA} + \beta_8 \text{PRICE}$$

$$V_3 = \beta_0 \text{NoBuy}$$

Price was the only variable that was coded numerically instead of categorically, the underlying assumption being that the marginal utility of money is constant over the range of expenditure, which we consider an acceptable approximation since the range of monetary variation is small.

Thus, the probability that individual i will choose steak in alternative j out of k alternatives in the choice task is:

$$\Pr(j) = \text{Prob}(U_{ij} > U_{ik}) \forall j \neq k$$

The assumption of independently and identically distributed (i.i.d.) Gumbel-distributed error term leads to a logit model[26] known as Multinomial Logit model (MNL), in this case the probability that the individual i will choose the alternative j is a logit probability:

$$P(Y = j) = \frac{e^{\beta' x_{ij}}}{\sum_{j=1}^{J} e^{\beta' x_{ij}}}$$

The unknown population parameters β are typically estimated by maximizing the sample likelihood over the parameter space. An MNL model imposes the condition that e_{ij} is i.i.d., while β does not vary across individuals (preference homogeneity) and all members of the population are preference 'clones'. However, these conditions are too restrictive in most applications. The random parameter MXL model for panel observations introduced by Boyd and Mellman[35] and popularized by Train[36] obviates the limitations of preference homogeneity (all respondents as preference clones) and allows for the more realistic hypothesis of taste variation across respondents. In an MXL model, the utility of individual i from alternative j is specified as

$$U_{ij} = \beta'_i X_{ij} + e_{ij}$$

where X_{ij} are observed variables that relate to each alternative and decision maker, β_i is a vector of coefficients for these variables for individual i representing that person's tastes and e_{ij} is a random term.

So, the utility function is defined as

$$U_{ij} = (\beta + \Omega' \eta_i)' X_{ij} + e_{ij},$$

where β is the mean vector and the idiosyncratic departures are $\Omega' \eta_i$ where the first term is a Cholesky matrix that introduces correlation across random effects, and suitable distributional assumptions are made for the random departure η_i.

Conditional on β_i, the likelihood that individual i chooses alternative j is still a logit probability. However, since β_i is random and not known, the (unconditional) choice probability is the integral of this logit formula over the density of β_i. This is approximated by simulation at the stage of estimation. The ultimate purpose of estimating a model of choice is generally to obtain unbiased estimates of the mean taste parameter vector β, and the pattern of variance–covariance of the idiosyncratic components Ω. Generally, in both MNL and MXL specifications, marginal WTP for each attribute of choice is calculated as the ratio of coefficients:

$$\text{WTP} = -\frac{\beta(\text{attribute level})}{\beta(\text{price})}$$

This way of proceeding employs models specified on the 'preference space', where often the price coefficient is held constant. This assumption makes the estimation easier but requires undesirable restrictions on the model, such as a constant marginal utility of money.

Instead of modeling taste heterogeneity in preference space—that is by specifying the distribution of coefficients in the utility function and then using them to derive the distribution of marginal WTP—Train and Weeks[37] proposed a way of specifying utility in the monetary space (WTP-space). This allows the researcher to directly assume the distributions of marginal WTP, which is very advantageous, while requiring a simple reparameterization. Recent studies have found that models in preference space tend to provide a less reasonable distribution of WTP than the models in WTP-space[37,38] even though at times they fit the data better. However, Scarpa and Rose,[29] in their application of WTP space models on environmental issues, found that models in WTP-space also fit the data better than the model in preference space. Similar results were obtained by Balcombe et al.[39].

One advantage of using panel models with repeated choices from each respondent is that one can obtain an estimate of WTP for each individual respondent by conditioning on the sequence of choices contexts for each. Such posterior, conditional estimates provide more accurate information for each respondent and facilitate validation.

Results

The data were analyzed by means of standard MNL model, random-parameter MXL models, both of which used preference space specifications, and then using random-parameter WTP space models (WTP-space). Both the random-parameter models were estimated under two specifications of the behavioral profile of preference heterogeneity for each modal attribute of the variables: a normal distribution and a constrained

Table 4. Estimated coefficients for the different models.

Variables	MNL	MXL (triangular)	MXL (normal)	WTP-space (triangular)	WTP-space (normal)
RED color	0.14***	0.21***	0.27***	2.68***	2.85***
FAT visible	−0.07*	−0.08	−0.13*	−0.98*	−0.98
ORGanic	0.97***	1.61***	1.96***	19.75***	17.73***
CONVentional	0.32***	0.31***	0.69***	5.26***	6.37***
FREE range	0.42***	0.64***	0.84***	8.13***	8.58***
LOCal breed	0.17***	0.22***	0.29***	3.05***	3.19***
ITAlian origin	0.62***	0.99***	1.18***	12.46***	12.62***
PRICE	−0.06***	−0.08***	−0.10***	1.00	1.00
NB (no buy)	−1.09***	−1.98***	−2.06***	−25.49***	−29.05***
sRED		0.21***	0.34***	2.68***	3.73***
sFAT		0.08	0.67***	0.98*	6.22***
sORG		1.61***	1.66***	19.75***	18.24***
sCONV		0.31***	0.63***	5.26***	3.27
sFREE		0.64***	0.71***	8.13***	8.34***
sLOC		0.22***	0.09	3.05***	0.9
sITA		0.99***	0.96***	12.46***	9.58***
sPRICE		0.08***	0.05***	–	–
sNB		2.25***	2.48***	26.96***	29.05***
Log-likelihood	−1499.18	−1256.99	−1202.06	−1256.99	−1202.06
Variables in model	9	10	18	11	18
Pseudo R^2	0.2134	0.342438	0.3679	0.3424332	0.3679

***, **, *Significance at 1%, 5% and 10% level.

triangular distribution. Following Hensher et al.[40] and Greene et al.[41], the constrained triangular distribution with mean parameter equal to its spread was chosen in order to achieve behaviorally plausible WTP distributions and bound large and implausible value estimates. We compare these results to those obtained assuming a normal distribution of taste heterogeneity. The no-buy option was specified as normally distributed in both cases.

Table 4 shows coefficient estimates for MNL, MXL and WTP-space models and Table 5 reports the coding of all variables. The last two models allowed for estimating individual-specific WTP for the selected beef steak attributes. All variables, apart from price and status quo, are effect coded. With effect coding the constant is equal to the grand mean of all of the observations. The coefficients of each of the effect variables are equal to the difference between the mean of the group coded 1 and the grand mean. Effect coding in a choice experiment allows the estimation of the effects of all levels without any correlation with the constant. However, when the variables are effect coded, the estimated unconditional and conditional WTPs have to be multiplied by 2 to obtain the actual marginal WTP values[42].

With few noticeable exceptions, standard deviations are significantly different from zero for the majority of variables in all models, showing heterogeneity in consumer preferences (Table 5). Generally speaking, the WTP estimates increase when going from MNL (with no taste heterogeneity) to WTP-space models. In both preference- and WTP-space, the specification with normally distributed taste fits the data better (Log-L −1202.06) than the one specified assuming a constrained triangular distribution for the parameters (Log-L −1256.99). Since the models in WTP-space provide a more reasonable distribution of individual WTPs than the models in preference space, we will concentrate on the results from the former.

In Table 6, we report the mean conditional WTP estimates for the attributes of interest in this study, namely the ethical and environmental attributes. The variable

Table 5. Variable and coding.

Alternative specific variables	Operalization
RED = color of the beef; 1 = red, −1 = pink	Effect coded
FAT = meat fat; 1 = visible fat, −1 = no visible fat	
ORG = beef from organic method; 1 = organic, −1 = not organic	
CONV = beef from conventional method; 1 = conventional, −1 = not conventional	
FREE = free range; 1 = free-range animals, −1 = animals in box	
LOC = breed origin; local breed = 1, international breed = −1	
ITA = place of production; Italy = 1, Abroad = −1	
PRICE = price for 1 kilo of beef meat in euro	Metric
NB = constant for no-buy option; = 1 if no-buy option was chosen, = 0 otherwise	Dummy coded

*GM-fed beef = (ORG = −1, CONV = −1).

Table 6. Mean conditional WTPs for ethical and environmental attributes.

	MXL (triangular distribution)					MXL (normal distribution)				
	CONV	DORG	FREE	LOC	ITA	CONV	DORG	FREE	LOC	ITA
Mean	10.15	40.63	20.94	7.32	32.62	19.59	41.35	19.88	7.21	30.57
SD	3.68	20.82	8.66	2.67	16.38	45.21	121.74	41.16	12.56	59.03
Min.	5.18	3.41	7.32	3.71	9.94	−122.85	−630.96	−116.27	−21.69	−131.90
Max.	24.36	129.13	66.98	18.26	91.91	499.38	1021.18	337.69	126.07	601.40

	WTP-space (triangular distribution)					WTP-space (normal distribution)				
	CONV	DORG	FREE	LOC	ITA	CONV	DORG	FREE	LOC	ITA
Mean	10.43	55.37	31.59	12.27	48.22	*12.76*	*26.25*	*17.29*	*6.40*	*24.69*
SD	0.98	22.46	6.28	1.11	12.73	*2.08*	*27.42*	*11.55*	*0.50*	*15.05*
Min.	7.20	5.78	12.20	8.99	18.53	*3.24*	*−52.83*	*−28.38*	*4.87*	*−15.43*
Max.	14.06	120.56	48.44	15.17	75.63	*18.02*	*92.01*	*44.10*	*8.01*	*59.98*

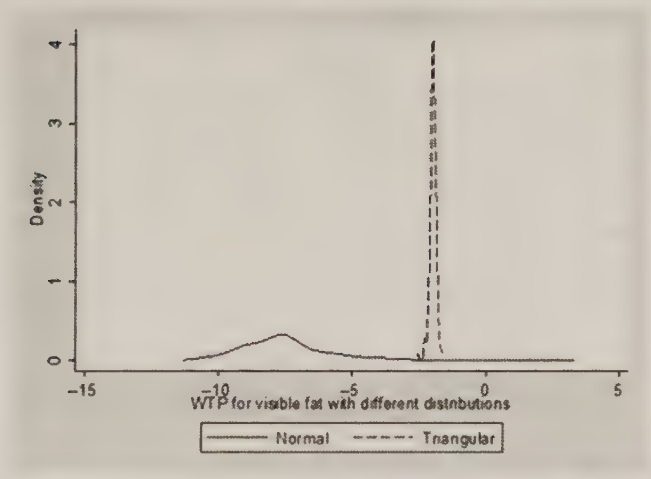

Figure 1. Kernel density plot for conditional WTP of FAT (triangular and normal distributions).

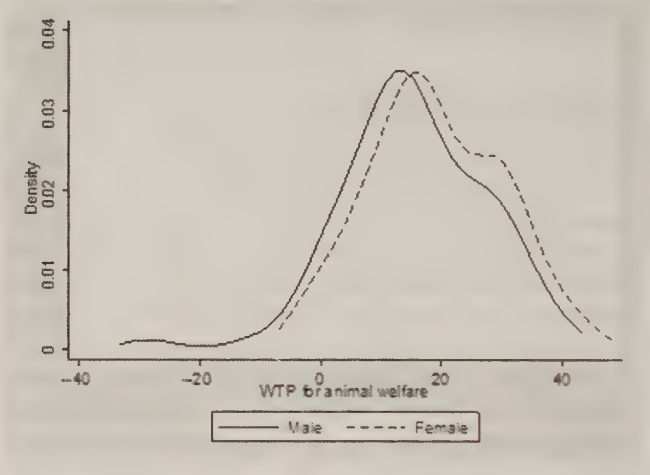

Figure 2. Kernel density plots of conditional WTP for animal welfare in different gender groups.

DORG consists of the mean difference in individual-specific WTP estimates between organic and conventional production, and approximates the 'premium price' per kg of beef derived from consumer choice statements. Hypothetical choices are known to lead to over-estimates and these amounts should probably be discounted by one-sixth to account for hypothetical bias[43].

The mean value estimate for visible fat is not different from zero for most of the models. However, the estimated standard deviation is significant, indicating that there is much variation across consumer preferences. Respondents are split into two distinguishable groups: people who do not like visible fat in beef steak (the majority) and a vast group of people who either do not like fat or are indifferent to it.

The tendency is clearly visible in Figure 1 where the distribution of individual-specific WTP for the fat attribute at individual level is reported (from the WTP-space model), for both normal distribution and constrained triangular. Using the latter distribution—which constrains the parameters in the 0–2β range—the heterogeneity in consumer tastes is reduced to a minimum, but this is an artificial result not representing the true spanning of preferences between the 'visible fat in steak' lovers and those who want lean meat. This shows a shortcoming of using bounded or constrained distributions. Therefore we will stick to the unconstrained, normally distributed WTP-space estimates as most reliable.

Socio-demographic characteristics in most cases do not seem to influence choices nor implied marginal values. The only noticeable exception is gender: women have a higher WTP for higher animal welfare than men, and the difference is statistically significant (Fig. 2).

Organic production and domestic origin were the most relevant attributes for respondents. Consumers reject GM-fed beef, which is perceived as a 'bad' good or non-good, in contrast to other types of beef. We estimated a

difference in WTP of 12.76 euros/kg between conventional and GM-fed beef. Thus, compared with a medium price of 12 euros/kg (average price in the three survey locations in euros/kg in February 2008, for a conventional fresh beef steak, first cut[44]), consumers are willing to *be paid* 0.76 euro/kg for purchasing GM-fed beef, confirming the fact that GM-food is something European (and, especially, Italian) consumers are not willing to accept without compensation.

Organic is perceived as a much safer production method to avoid GM contamination risks than conventional. Besides, organic entails other environmental- and health-related meanings[8]. Looking at the normally distributed, WTP-space estimates in Table 6 (those in *italics*), consumers are WTP a price premium (DORG = ORG − CONV) of 26.25 euros/kg. Even considering the correction for hypothetical bias (one-sixth of the marginal WTP), the resulting absolute WTP (12 + 21.86 = 33.88 euros/kg) is well above current average organic market price (24 euros/kg). An explanation for such a high result is twofold. First, since our sample included highly educated and older-than-average Italians, a portion of this premium could be ascribed to an (unmeasured) income effect. Second, one must consider that the European (and Italian) market for organic beef is very volatile: periods of undersupply follow periods of oversupply, due to the fact that high WTP attracts new farmers but then—due to the time needed for conversion—time is required to react to market price increases. It happens that prices tend to increase further attracting farmers into the market until the market drops[1,45]. In 2008, there was a period of undersupply in the Italian organic beef market[46]. So, the results may indicate that the market did not fully exploit the potential premiums. Indeed, such a fact may be explained by a long-term strategic approach to marketing of organic beef: increasing organic prices in the short term would allow market equilibrium (since some consumers will simply stop demanding the product), but will eventually hamper the future prospect of the sector, when supply would eventually increase. So, for retailers, it is probably better to accept a higher amount of unsatisfied demand than to accept sharp rises of organic beef price.

Other ethical and environmental variables are also valued positively: estimates ranges from 24.69 euros/kg for lower food miles (i.e., national origin: ITA) to 6.40 euros/kg for biodiversity preservation (i.e., LOCal breeds), passing through 17.29 euros/kg for animal welfare (FREE). Globally, ethical and environmental issues (credence cues) appear more relevant in explaining the overall WTP for organic beef than ordinary product characteristics (search cues: RED, FAT).

Concluding Remarks

Albeit our study was limited in geographic scope and sample representativeness, we conclude that ethical/environmental issues (credence cues) appear to play a relevant role in explaining WTP variation for organic beef, perhaps more so than ordinary product characteristics (search cues). For example, our results confirm that animal welfare is a relevant extrinsic cue in organic meat purchase, coupled with local/nation origin and local breeds[47–52]. These preliminary results suggest a potential increase in the role of ethical concerns in consumers of organic beef in Southern European countries (such as Italy), where health-related credence attributes used to be the main drivers for organic consumption[8,11].

Further research should address some of the limitations of our study, in order to confirm or disconfirm our findings.

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Reconciling productivity with protection of the environment: Is temperate agroforestry the answer?

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Abstract

Meeting the needs for a growing world population calls for multifunctional land use, which can meet the multiple demands of food and fuel production, environmental and biodiversity protection, and has the capacity for adaptation or resilience to climate change. Agroforestry, a land-use system that integrates trees and shrubs with crops and/or livestock production, has been identified by the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD) as a 'win–win' approach that balances the production of commodities (food, feed, fuel, fiber, etc.) with non-commodity outputs such as environmental protection and cultural and landscape amenities. Evidence is now coming to light that supports the promotion of agroforestry in temperate developed countries as a sustainable alternative to the highly industrialized agricultural model with its associated negative environmental externalities. This paper reviews this evidence within the 'ecosystem services' framework to evaluate agroforestry as part of a multifunctional working landscape in temperate regions. Establishing trees on agricultural land can help to mitigate many of the negative impacts of agriculture, for example by regulating soil, water and air quality, supporting biodiversity, reducing inputs by natural regulation of pests and more efficient nutrient cycling, and by modifying local and global climates. The challenge now lies in promoting the adoption of agroforestry as a mainstream land use through research, dissemination of information and policy changes.

Key words: ecosystem services, sustainability, trees, silvoarable, silvopastoral, carbon sequestration

Introduction

The Food and Agriculture Organization (FAO) predicts that food production (net of food used for biofuels) must increase by 70% by 2050 to meet the needs of a growing world population[1]. A major concern is that this will put greater pressure on marginal land and protected habitats, plus intensify existing agricultural practices, with serious implications for the ecological processes that sustain human well-being ('ecosystem services'[2]). This food/fuel/biodiversity conflict calls for multifunctional land use, which can meet the multiple demands of food and fuel production, environmental and biodiversity protection, and has the capacity for adaptation or resilience to climate change.

One approach is to design farming systems that mimic the structure and function of natural ecosystems, based on the hypothesis that natural systems are eco-efficient, with internal cycling of nutrients and energy and protection of the resource base. Agroforestry, a land-use system that integrates trees and shrubs with crops and/or livestock production, builds on this idea of ecological design to optimize beneficial interactions between the woody and other components. A central hypothesis in agroforestry is that productivity is higher in agroforestry systems compared to monocropping systems due to complementarity in resource-capture, i.e., trees acquire resources that the crops alone would not[3].

In addition to increased productivity, agroforestry systems are believed to provide a number of other ecosystem services and environmental benefits, including biodiversity conservation, regulation of soil, air and water quality and carbon sequestration[4]. Agroforestry has been identified by the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD) as a 'win–win' multifunctional land-use approach that balances the production of commodities (food, feed, fuel, fiber, etc.) with non-commodity outputs such as environmental protection and cultural and landscape amenities[5]. While much of the

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research has focused on agroforestry in tropical systems as a means of addressing both food security and environmental issues in impoverished rural areas, evidence is now coming to light that supports the promotion of agroforestry in temperate developed countries as a sustainable alternative to the highly industrialized agricultural model with its associated negative environmental externalities. In the following review, we examine this evidence in the context of the 'ecosystem services' framework to evaluate agroforestry as part of a multifunctional working landscape in temperate regions.

Definition and classification

While systems integrating trees and agriculture have been practiced for 1000s of years, the term 'agroforestry' was first coined in 1977, in a review of research needs, *Trees, Food and People*, by Bene et al.[6]. In its simplest form, agroforestry can be defined as 'growing trees on farms'[7]. It is generally accepted, however, that agroforestry systems are deliberately designed and managed to maximize positive interactions between tree and non-tree components. A widely accepted definition was formulated by Lundgren in 1982:

> Agroforestry is a collective name for land-use systems in which woody perennials (trees, shrubs, etc.) are grown in association with herbaceous plants (crops, pastures) or livestock, in a spatial arrangement, a rotation, or both; there are usually both ecological and economic interactions between the trees and other components of the system[8].

This represents a concept of integrated land use that combines elements of agriculture and forestry in a sustainable production system. The emphasis here is on managing rather than reducing complexity. Agroforestry uses the natural woodland ecosystem as a model to create 'a dynamic, ecologically-based, natural resources management system'[9]. Agroforestry systems can be classified according to the components present—trees with crops are referred to as silvoarable, trees and animals as silvopastoral, and trees with crops and animals as agrosilvopastoral[10].

Ecosystem Services

Agroforestry is valued as a multifunctional approach that balances productive and protective functions, therefore avoiding the trade-off between provisioning and other ecosystem services that occurs in many modern intensive farming systems.

Provisioning services

Agroforestry supports the production of a wide range of products due to the diversity of planned species within the system (Table 1). Higher productivity from agroforestry systems is based on the ecological theory of niche differentiation; different species obtain resources from different parts of the environment. Tree roots generally extend deeper than crop roots, and so access soil nutrients and water unavailable to crops, as well as absorbing nutrients leached from the crop rhizosphere[11]. These nutrients are then recycled via leaf fall onto the soil surface or fine root turnover. This should lead to greater nutrient capture and higher yields by the integrated tree–crop system compared to tree or crop monocultures. In addition to greater resource capture, increased overall productivity of agroforestry systems also occurs as a result of modification of the microclimate by the presence of trees, which benefits crop growth and livestock production[12]. The establishment of trees on land deemed unsuitable for productive cropping is another way that agroforestry can increase overall productivity from farming systems. Dixon et al.[13] estimated that 140 million ha of marginal or degraded land in North America was available and suitable for agroforestry establishment. In New Zealand, trees have been re-introduced into pasturelands to reduce soil erosion on poor soils; the productive potential of these trees is now being recognized and exploited either for timber production or as a feed resource[14].

Table 1. Overview of products from agroforestry systems.

Overview of products from agroforestry systems
Food: Depending on the design of the system, food products can include arable crops, vegetables, animal products (meat, eggs, dairy), fruit, mushrooms, oils, nuts and leaves
Fuel: Agroforestry systems can be managed to generate fuel wood as its primary woody product (e.g., willow or hazel coppice, charcoal) or as a by-product of management for other products (e.g., prunings of side branches)
Fodder and forage: Livestock can browse on tree leaves and forage grasses and legumes *in situ* or fodder and hay or silage can be cut, transported and stored
Fiber including wood pulp for paper, cedar pet bedding, pine straw, rubber, cork, bark and woodchip mulch, as well as animal fibers such as hair, fleece, horn and leather and silk from silk moths and herbaceous plant fibers (e.g., cotton, linen and flax)
Timber production for furniture, construction, musical instruments, veneers and shipbuilding
Gums and resins for varnishes (copal, dammars, mastic and sandarac), adhesives, incense and perfume (e.g., frankincense and myrrh)
Thatching and hedging materials, including spars, binders and stakes
Gardening materials including pea sticks, bean poles, fencing and hurdles
Medicinal products including ginseng, goldenseal and witch hazel
Craft products such as natural dyes, wooden carvings, basketry, weaving and sculptures, floral arrangements and decorations, including Christmas trees, holly, mistletoe and ivy
Recreation: fee hunting (for deer, gamebirds, etc.) and agritourism (aesthetic backdrop for horse-riding, fishing, wildlife watching and holiday parks)

Interactions between trees, crops and livestock. Ecological interactions between the tree and crop or animal components in an agroforest occur above ground (microclimate effects) and below ground (root interactions), and involve a number of complex processes related to soil fertility, competition, microclimate, pests and diseases, soil conservation and allelopathy[12,15]. Interactions may be direct (e.g., cattle browsing on leaves) or indirect via changes in the environment (e.g., soil compaction caused by cattle gathering in the tree shade, reducing tree growth)[11] and occur at multiple scales. The strongest interactions are likely to occur in the zone immediately surrounding the tree (the tree–crop interface), while shelter effects are known to extend up to 30 times the height of the tree[16,17]. The influence of trees on soil and hydrological processes may impact watershed/catchment scale systems[18,19], biodiversity benefits may be recorded at the landscape scale, and carbon sequestration effects at a global scale[4].

As agroforestry systems are dynamic, these interactions are likely to change over time, so that there may be complementarity between the components in the early stages, which then shifts into competition for resources as the tree component reaches maturity[12]. The nature of interactions also vary depending on different short- and medium-term ecological factors, such as seasonal and annual variations in rainfall and temperature, so that the balance between the tree and the crop can swing between positive interactions (facilitation) and negative interactions (competition)[20]. The productivity of a system is a net result of these interactions[12].

Measuring productivity. The land equivalent ratio (LER), first proposed by Mead and Willey[21], is a means of comparing productivity of intercropping and monocropping systems. It is calculated as the ratio of the area needed under sole cropping to the area of intercropping at the same management level to obtain a particular yield:

$$\mathrm{LER} = \frac{\text{Tree agroforestry yield}}{\text{Tree monoculture yield}} + \frac{\text{Crop or livestock agroforestry yield}}{\text{Crop or livestock monoculture yield}}.$$

If a rotation includes more than one crop, a weighted ratio for each crop can be used, based on its proportion in the rotation. An LER of 1 indicates that there is no yield advantage of the intercrop compared to the monocrop, while an LER of 1.1 indicates a 10% yield advantage, i.e., under monocultures, 10% more land would be needed to match yields from intercropping[22]. Yields can be expressed in physical units so that the LER refers to the biological efficiency of the mixture, or monetary units where the LER indicates the economical efficiency of the mixture. Biomass yield LER's of agroforestry systems range from 2 in a pear orchard/radish system (Newman, 1986, in[22]) to 1.6 in the early years after establishment of a cherry/fescue system, declining to 1.0 later in the rotation, with an average of 1.2 over the 60-year rotation[22].

Economics. Economic studies of agroforestry systems have shown that financial benefits are a consequence of increasing the diversity and productivity of the systems, influenced by market and price fluctuations of timber, livestock and crops[23–27]. Incorporating trees into an agricultural system produces higher long-term returns (Thorrold et al., 1997, in[28]), while agroforestry practices are able to recoup initial costs more quickly than forestry-only land use due to the income generated from the agricultural component[26]. Fernández-Núnez et al. (2007, in[26]) carried out an assessment of initial investments and establishment costs of forestry, agriculture and agroforestry in the Atlantic area of Spain. They found that establishing agroforestry required higher initial investment than the agricultural and forestry systems due to higher initial inputs, but over a 30-year period, profitability per hectare was higher in the agroforestry system than in the exclusively livestock (17%) or forestry (53%) systems.

Modeling of economic returns from a black walnut alley cropping system in Midwestern USA highlighted the importance of system design and management for maximizing productivity[23]. Systems with widely spaced tree rows (12.2 m between tree rows) were predicted to be more profitable than both closely spaced (8.5 m between tree rows) designs and walnut plantations, while root-pruning increased economic returns by extending the period of profitable crop production within the rotation. All agroforestry systems were modeled as having higher rates of returns than monocropping systems[23].

Regulating services

Air quality regulation. Trees have long been used in urban settings to take advantage of their capacity to ameliorate air pollution from vehicles, buildings and industry. The use of agroforestry in mitigating livestock odor has recently been explored by Tyndall and Colletti[29]. Intensive livestock production units in the American Midwest emit odors that have caused significant social problems with implications for rural and state economies, human health and the quality of rural life. The situation is exacerbated by the highly modified landscapes that lack natural barriers that surround these units, which allow emissions to travel further. Tyndall and Colletti[29] recommend the use of shelterbelts to biophysically mitigate odors, with shelterbelts of modest heights ideal for intercepting the movement of particulates which form the majority of odors.

Climate regulation. Integrating trees into the agricultural landscape can regulate the climate at the local scale through modification of the microclimate and at the global scale through reductions in atmospheric carbon by sequestering, conserving and substituting carbon.

Microclimate modification. As 'ecosystem engineers' (i.e., they directly and indirectly influence ecosystem dynamics by changing the availability of abiotic and biotic resources to other species in an ecosystem[30,31]), trees modify microclimatic conditions, including temperature, humidity and wind speed, which can have beneficial effects on crop growth or animal welfare[12].

Trees provide an important service by reducing wind speed with reductions extending up to 30 times the height of the windbreak on the leeward side, and between two and five times the shelterbelt height on the windward side[16,17]. This can have multiple benefits for crops; increased growth rate and quality, protection from windblown soil, moisture management and soil protection[12,16,28,32–35]. Higher air and soil temperatures in the lee of a shelterbelt can extend the growing season, with earlier germination and more growth at the start of the season[33]. Fruit and vegetable crops are particularly sensitive to wind stress and suffer lower yields and poorer quality at lower wind speeds than agronomic crops[16,33]. Horticultural crops grown in the shelter of a wind break benefit from moderation of temperature extremes, warmer soil and air temperatures, and improved soil water conditions, all of which contribute to increasing total marketable yields and individual fruit weight[33]. Sheltered conditions increase flowering periods and bee activity, leading to increased fruit set and earlier maturity[36].

Shading effects of trees may result in higher levels of soil water content beneath the canopy compared to open cropland or pasture due to tree shade reducing evapotranspiration, especially in semi-arid regions[28]. In Spanish *dehesas*, shade from the cork oaks significantly improved the microclimate of a fodder crop (oats), with a reduction in daily and seasonal variation in soil and air temperatures in the proximity of the trees having implications for understorey phenology and physiology and lengthening the growing season[34].

However, where the tree and crop or livestock components overlap in their use of resources, competition may lead to reduced productivity compared to a monoculture system. Within northern temperate regions, the main limiting resource for plants is usually light, and studies have shown that shading has reduced yields in temperate agroforestry systems[28,37,38]. Competition for water between tree and crop components is likely to limit productivity in semiarid regions such as the Mediterranean, although it is difficult to separate competition for water from that for nutrients[12] and, indeed, reduced evapotranspiration due to tree shade effects on understorey plants may increase soil water content compared to open pastures[39]. The complex relationship between soil water content, rainfall, water uptake by plants and evapotranspiration throughout the seasons makes it extremely difficult to fully understand water dynamics within an agroforestry system. In addition to competing for resources, some species of plants and fungi can have a direct negative impact on others through the production of biochemicals called allelochemicals that influence germination, growth, development, reproduction and distribution of other organisms. These allelochemicals can be released into the rhizosphere as plant residues decompose or via root exudates[12]. For example, walnut and pecan trees produce juglone, a phenolic compound that has been shown to inhibit survival and growth of several herbaceous and woody plants in pot experiments[40].

Shelter for livestock. Modification of the microclimate by trees provides many benefits for livestock; trees provide shelter from rain and wind, shade from the sun, and cover from predators. Cattle are particularly sensitive to heat stress and evaporative cooling is the primary mechanism by which cattle reduce their temperature. By providing shade, trees can reduce the energy needed for regulating body temperatures, and so result in higher feed conversion and weight gain. Research in the southern United States found that cattle that had been provided with shade reached their target weight 20 days before those with no shade[41]. Higher respiration rates and lower activity rates of those cattle without shade were thought to reduce productivity[42]. Cattle are more evenly distributed in the silvopasture compared to open pasture, which has important implications for optimizing forage use and uniform nutrient recycling[42].

During cooler months, windbreaks and shelterbelts provide valuable protection from the wind for livestock, particularly for new-born lambs and freshly shorn sheep. When livestock have been protected from winter storms by windbreaks, significant savings in feed costs, survival and milk production have been reported by producers in Dakota, US (Anderson and Bird, 1993, in[33]).

For livestock species such as pigs and poultry that have evolved from forest-dwelling ancestors, including tree cover within their range can influence ranging behavior and enhance animal welfare. For poultry, trees offer protection from aerial predators in particular, and can provide an escape from aggressive behavior within the flock, as well as reducing visual stimulation that can provoke aggression[27]. Dawkins et al.[43] observed ranging behavior in commercial free-range broiler (table bird) systems and found that the number of birds ranging outside was correlated with the percentage tree cover on the range, with behavioral studies showing that trees and bushes were the preferred habitat[43]. Like chickens, pigs have a forest-dwelling ancestor, the Eurasian wild boar (*Sus scrofa scrofa*), which is found primarily in mixed, predominantly deciduous woodland. Behavioral studies of domestic pigs have shown that trees encourage expression of normal behavioral patterns[44].

The inclusion of a diverse range of woody species within livestock range has potential for enhancing animal health and welfare by providing opportunities for self-medication. For example, willow has been traditionally

used as a feed resource for its anti-helminthic properties[45]. However, there has been little recent research exploring the full potential of this approach.

Global climate. There has been an increase in research over the past 20 years investigating the potential of agroforestry as a tool for addressing the issues of climate change through mitigation and adaptation[46–51]. Agroforestry has the potential to reduce atmospheric carbon, by increasing afforestation of agricultural lands (C sequestration), by reducing resource use pressure on existing forests (C conservation) and by producing both durable wood products and renewable energy resources (C substitution). However, agroforestry systems can also be sources of C and other greenhouse gases, where tillage, burning, manuring, chemical fertilization and frequent disturbance can result in emissions of CO_2, CH_4 and N_2O into the atmosphere[52]. Establishing agroforestry systems on previously natural primary or secondary forests will reduce C sequestration potential, whereas planting trees on degraded or treeless agricultural land will increase their C sequestration value[49].

C sequestration. Agroforestry can increase the amount of carbon sequestered compared to monocultures of crops or pasture due to the incorporation of trees and shrubs[4]. The potential for agroforestry systems to sequester carbon depends on a number of factors, including system design, species composition and age, environmental factors such as climate, management and the end product. As a result, estimations for carbon sequestration of agroforestry systems vary from $0.29\,Mg\,ha^{-1}\,yr^{-1}$ in a fodder bank system in West Africa to $15.21\,Mg\,ha^{-1}\,yr^{-1}$ in mixed species stands in Puerto Rico[53]. Schroeder[46] estimated average carbon storage by agroforestry systems as 9, 21, 50 and $63\,Mg\,C\,ha^{-1}\,yr^{-1}$ in semiarid, sub-humid, humid and temperate regions, respectively, with higher rates in temperate regions reflecting longer rotations and longer-term storage.

Carbon substitution. Biomass energy from short rotation coppice (SRC) is a carbon-neutral source of energy that does not contribute to CO_2 enrichment of the atmosphere. SRC woody crops such as willow produce between 11 and 16 units of useable energy per unit of non-renewable fossil fuel energy used to grow, harvest and deliver SRC[54]. However, there have been concerns that widespread adoption of biomass crops such as *Miscanthus* and SRC willow will compete with food production and impact biodiversity[55,56]. Incorporating SRC into an agroforestry system is one approach to reconciling these conflicting demands. In temperate regions, species with potential as SRC's include poplar (*Populus* spp.), willow (*Salix* spp.) and black locust (*Robinia pseudoacacia*). SRC species such as willow are often suitable for unproductive land such as erodible land or waterlogged sites, thus providing income from marginal land, and as SRC is usually harvested on a 3-year cycle, it provides a more regular income than other forms of agroforestry (e.g., timber).

Greenhouse gas abatement. The role of temperate agroforestry in mitigating greenhouse gases has not yet been investigated fully, although a review of tropical systems highlights the potential of agroforestry for mitigating CO_2 and N_2O and increasing the CH_4 sink strength compared to monoculture systems[57]. In the UK, current work by the Centre for Ecology and Hydrology in Edinburgh is exploring the potential of farm woodlands for ammonia abatement using targeted field measurements and mechanistic and atmospheric emission modeling[58]. In agroforestry systems, there is a reduced need for supplementary nitrogen applications, and recycled nitrogen from leaf litter provides a quantifiable contribution to adjacent crops that can replace inorganic N additions and thus reduce N_2O emissions[59]. A decrease in nitrogen leaching out of the rooting zone will reduce NO_x emissions as a result of denitrification in surface water resources[59]. Models estimate that nitrates leaving a tree-based intercropping system can be reduced by 50% compared to a monoculture control[59].

Water quality and regulation. Research has demonstrated that agroforestry can reduce pollution from crops and grazed pastures[60–64]. Riparian (riverside) buffers in particular, can reduce non-point source water pollution from agricultural land in five general ways[65]:

1. Reducing surface runoff from fields.
2. Filter surface runoff.
3. Filter groundwater runoff.
4. Reduce bank erosion.
5. Filter stream water.

The 'safety net hypothesis' is based on the belief that the deeper-rooting tree component of an agroforestry system will be able to intercept nutrients leached out of the crop-rooting zone, thus reducing pollution and, by recycling nutrients as leaf litter and root decomposition, increasing nutrient use efficiencies[12]. Greater permanence of tree roots means that nutrients are captured before a field crop has been planted and following harvest, when leaching may be greater from bare soil.

Buffer strips can significantly decrease pollution run-off, with reductions of 70–97% reported for suspended solids, 60–98% for phosphorus and 70–95% for nitrogen[62,66]. Agroforestry systems also have the potential to mitigate movement of harmful bacteria such as *Escherichia coli* into water sources[60] and reduce the transport of veterinary antibiotics from manure-treated agroecosystems to surface water resources[67]. Agroforestry has been used to address issues of soil salinization in Australia where a study recorded a lowering of the saline groundwater table by 2.0 m over a 7-year period under a *Eucalyptus*–pasture system, relative to nearby pasture-only sites[68].

A principal cause of non-point source pollution and soil erosion is excessive surface water runoff. Riparian (river bank) buffers and other agroforestry systems can help reduce runoff and increase infiltration[18,19,61]. In Midwestern USA, a multispecies buffer that included

woody perennials increased infiltration rates to five times that of cultivated and grazed fields[19]. Agroforestry can reduce soil water content during critical times such as fallow periods and increase water infiltration and water storage. Furthermore, above ground, stems, leaf litter and pruning debris in agroforestry systems can reduce runoff flow rates, thereby enhancing sedimentation within the agroforestry strip and increasing infiltration[19].

The role of agroforestry in rehabilitating polluted soils has been investigated, through exploiting the ability of trees to capture nutrients and pollutants. For example, research has shown that willows can take up heavy metals from soil into their biomass, help break down pollutants to non-toxic compounds and control water dynamics, including contaminated groundwater flow and water penetration into soils via evapotranspiration[54]. Agroforestry systems have been used to recycle urban and agricultural organic waste with the added benefit of increased biomass productivity from the additional nutrients[69,70]. Previously a burden to society, these waste products can be viewed as a valuable resource to maximize biomass production[71].

Erosion regulation. The conversion of natural forest and scrublands to grasslands devoid of trees by European settlers on hill country with steep slopes and erodible soils in New Zealand resulted in increased run-off and accelerated erosion[72,73]. Soil conservation measures involving farm woodlots and wide-spaced tree plantings are needed to support sustainable pastoral and cropping uses on 32% of the North Island and 25% of the South Island (Eyles et al., 1992, in[72]). New Zealand government-subsidized erosion control schemes in the 1960s and 1970s planted over 2 million poplars; during the late 1980s and 1990s there was a shift to planting schemes using radiata pine (*Pinus radiata*), poplar silvopastoral systems and riparian schemes using poplar and willows[72]. Willow and poplar are widely used on unstable hillsides in silvopastoral systems for many reasons: they are easy to propagate and establish; have fast early growth rates; are tolerant of flooding and partially saturated soils; have extensive root systems with fine fibrous root mats that stabilize large soil masses; have high evapotranspiration rates that can help dry out swampy soils; provide shade, shelter and fodder for livestock; and may produce useful timber if managed appropriately (poplar only)[72].

Pest and disease regulation. Reduced pest and disease problems in agroforestry systems are predicted due to greater niche diversity and complexity than in monocropping systems[74]. This can be attributed to a number of mechanisms[75]:

- Intermittent distribution of host plants makes it more difficult for pests to find the plants.
- One plant species acting as a trap-crop, which protects the other crop from herbivore attack.
- One plant species acting as a repellent to the herbivore.
- Higher predator and parasitoid densities due to higher plant diversity.
- Increased interspecific competition between pest and non-pest species.

Trees provide greater structural and microclimate diversity, greater temporal stability, greater biomass and surface area, alternate sources of pollen, nectar and prey, alternate hosts and stable refuges[74,76,77]. Agroforestry systems can be managed to enhance pest regulation, for example by providing sources of adult parasitoid food (e.g., flowers), and sites for mating, oviposition and resting sites[7,74].

Higher insect diversities and natural enemy abundance, and lower abundances of pest species have been recorded in several agroforestry systems[59,78–86]. For example, in an alley cropping system with peas (*Pisum sativa*) and four tree species (*Juglans, Platanus, Fraxinus* and *Prunus*) in Leeds, UK, higher insect diversities and natural enemy abundance and lower abundances of pea and bean weevils (*Sitona* spp.) and pea midge (*Contarinia pisi*) were recorded compared to a monoculture of peas[82]. In this same silvoarable system, grain aphids (*Sitobion avenae*) populations in the winter barley crop were approximately half that of the arable control[81]. This was attributed to an increase in cereal aphid predators, primarily hoverflies (Diptera: Syrphidae), which used the tree-strips as a refuge[83].

While most focus has been on beneficial effects of trees on associated crops and livestock, the tree component may also benefit from pest suppression by livestock. In a study of farmed wild boar in Norway Spruce plantations in Austria, it was observed that mortality rates of the sawfly *Cephalcia abietis*, a defoliant whose larvae pupate in the soil for 21–33 months, increased by an additional 35.2% due to predation and soil disturbance by the boar[87]. Similarly, local domestic breed pigs in South African plantations of *Pinus patula* killed 98% of the soil pupae population of the pest *Nudaurelia cytherea* (pine emperor moth) and 90% of a *Pseudobunaea irius* (poplar emperor moth) infestation (Van Den Berg, 1973, in[24]).

However, some pest groups have been observed in higher numbers in agroforestry systems, and shifts in relative importance of pest groups may present novel management problems and influence crop choice. For example, Griffiths et al.[88] observed increased slug populations and slug damage in agroforestry plots compared to arable controls in a silvoarable experiment in West Yorkshire, UK. This was attributed to the provision of refugia for slugs within the permanent, unploughed vegetated areas under the tree rows and a modified microclimate, which favor slug populations.

Biodiversity regulation. As ecosystem engineers and keystone species, trees provide a range of resources to other flora and fauna, and when integrated into agroecosystems, they can enhance structural heterogeneity and habitat stability of the landscape[89]. Jose[4] identifies five

main ways that agroforestry contributes to the preservation of biodiversity:

1. By providing habitat for species that can tolerate a certain level of disturbance.
2. By helping to preserve germplasm of sensitive species.
3. By helping to reduce the rates of conversion of natural habitat and alleviate resource use pressure.
4. By providing connectivity by creating corridors between habitat remnants and the conservation of area-sensitive floral and faunal species.
5. By helping to conserve biological diversity by providing other ecosystem services such as erosion control and water recharge, thus preventing habitat degradation and loss.

There have been a number of studies investigating the role of agroforestry in supporting biodiversity, especially in the tropics[90,91]. These studies demonstrate that agroforestry systems support floral and faunal assemblages that can be as species-rich, abundant and diverse as forests, but often with modified species compositions that include non-forest species[92]. While agroforestry systems are unlikely to provide habitat for specialist forest species that require large tracts of undisturbed forest or woodland, they can support biodiversity in otherwise open landscapes and allow movement of species between habitat remnants, as well as buffer protected areas from the impacts of more intensive systems[90].

Several studies have shown the value of temperate woodland edge habitats such as windbreaks, hedgerows and other agroforestry systems for invertebrate and vertebrate biodiversity, particularly in landscapes otherwise devoid of wooded habitats[93–96]. The impact of agroforestry on biodiversity may extend beyond the landscape-scale; Perfecto et al.[97] consider the correlation between decreasing populations of songbirds in eastern USA and the elimination of shade trees from coffee agroforests in Latin American countries. Those species in decline were migratory species that overwintered in the southern countries, and were found in the forest-like habitats of traditional coffee farms with a diversity of shade tree species.

Natural hazards and extreme events. With the prediction of an increase in extreme events such as flooding and droughts, increasing the number of trees within the landscape can act as buffers to reduce the impact on the farming system and wider environment. By reducing surface runoff and increasing infiltration and soil water holding capacity, the risk of flash flooding following periods of heavy rainfall is reduced in agroforestry systems, with the tree roots and trunks acting as permeable barriers to reduce sediment and debris loading into rivers following floods. In New Zealand, widely spaced poplars reduced pasture production losses due to landslides during a cyclonic storm by 13.8% with, on average, each tree saving 8.4 m^2 from failure (Hawley and Dymond, 1988, in[28]). Mature willow and poplar trees at 12 m spacing can reduce mass movement by 10–20% (Hicks, 1995, in[28]).

Cultural services

Cultural heritage values. Cultural aspects of agroforestry systems, particularly in temperate regions, are often overlooked, despite the long tradition of systems such as woodland and orchard grazing, alpine wooded pastures, pannage, the *dehesa* and parklands[98]. Lifestyles such as nomadism, transhumance and traditional techniques such as pollarding and hedge-laying, are integrated within such systems and the symbolic and cultural perception of these landscapes are shaped by local practices, laws and customs[99].

Societal benefits. A key objective of implementing agroforestry systems in the tropics is improving livelihoods of poor rural smallholders[100]. However, the societal benefits of temperate agroforestry have received less attention, with the focus limited primarily to economics[101]. Integrating trees into the agricultural landscape has the potential to impact the local economy through diversification of local products, diversification of rural skills, improvements to the environment, landscape diversification and economic stability. Wood products such as wood fuel (either as logs or wood chips) need to be produced in close proximity to end-users to reduce transportation costs and make the business economically viable. This creates important links and business relationships between the end-user and local community businesses, so that the money that is paid to obtain these products is spent locally, thus stimulating the local economy[54].

Recreation and ecotourism. Agroforestry systems can provide recreational opportunities that can benefit the general public, as well as the landowner. Activities such as hunting, fishing, mountain biking, equestrianism and rural tourism can diversify income for farmers, while the public can benefit from improved health and enjoyment from agroforestry through sports and wildlife watching[98]. Cultural landscapes such as the *dehesas* of Spain and Portugal, and the wood pastures of the Alps, can create financial opportunities through ecotourism.

Aesthetic values. Public attitudes to agroforestry reflect society's view of non-market benefits connected with amenity, habitat, landscape and animal welfare. The visual impact of monocultures of crops or trees is unappealing for many people; integrating trees into agricultural landscapes can increase the diversity and attractiveness of the landscape[98].

Supporting services

Soil formation and nutrient cycling. By promoting a closed system with internal recycling of nutrients, whereby nutrients are accessed from lower soil horizons by tree roots and returned to the soil through leaf fall, agroforestry systems enhance soil nutrient pools and turnover and reduce reliance on external inputs. For example, leaf fall from 6-year-old poplars resulted in mean soil nitrate

production rates in the adjacent crop–alley up to double that compared to soils 8.0–15.0 m from the tree row, and nitrogen release from poplar leaf litter was equivalent to 7 kg N ha^{-1} yr^{-1} [59]. Trees can also significantly influence nutrient additions to adjacent alley crops through intercepting rainfall, via throughfall (rainwater falling through tree canopies) and stemflow (rainwater falling down branches and stems). Zhang (1999, in[59]) showed that these pathways contributed 10.99 and 15.22 kg N ha^{-1} yr^{-1} in hybrid poplar and silver maple systems, respectively.

There have been many studies assessing the value of green mulch from leguminous trees to enhance soil fertility for adjacent crops in tropical agroforestry systems (e.g.,[102]). However, relatively few of the 650 woody species that are able to fix atmospheric nitrogen occur in temperate regions; of these black locust (*Robinia*), mesquites (*Prosopis*), alder (*Alnus*) and oleaster (*Eleagnus*) have been investigated for their nitrogen-fixing potential[12]. Significant transfer of fixed nitrogen to crops has been observed in a study, which showed that 32–58% of the total nitrogen in alley-cropped maize came from nitrogen fixed by the adjacent red alder (*Alnus rubra*)[12].

As many soil biological processes are performed by soil microorganisms, the presence of an abundant and diverse soil microbial community is essential to sustain productivity of an agroecosystem. In agroforestry systems, differences in litter quality between the tree and crop components promote spatial diversity in enzyme activities and microbial functioning and this spatial variation is enhanced by tree effects on microclimate[103]. Several studies have recorded higher microbial diversity, increased enzyme activity and greater stability in agroforestry alley cropping systems, attributable to differences in litter quality and quantity and root exudates[103–107].

Arbuscular mycorrhizal (AM) fungi enhance plant nutrient uptake and growth, soil stability and soil aggregation, litter decomposition rates, and could potentially enhance crop yields while reducing the need for chemical fertilizer input[108–110]. However, while AM fungal diversity tends to be low in conventionally managed agricultural soils, which has been attributed to negative effects of fertilization, fungicides, soil cultivations and low host diversity, it has been shown that agroforestry systems may enhance AM fungal richness compared to monocropped systems[111]. The role of AM symbioses in temperate regions have so far only been studied in intensive, high-input agroforestry; the potential of AM fungi to enhance plant growth in low-input and organic systems still needs quantifying[111].

Higher levels of soil organic matter in agroforestry systems also positively influence soil invertebrate communities[112,113]. In a poplar–arable silvoarable system, soil organic matter, soil arthropod abundance and cumulative body mass were higher in samples taken close to the trees, with lower levels in the crop alleys attributed to frequent cultivations, lower litter inputs and a reduction in tree root densities[112].

Valuing ecosystem services in agroforestry

Recently, there has been considerable interest in placing a monetary value on the delivery of ecosystem services or public goods, such as soil protection and carbon sequestration. Porter et al.[114] calculated the values of market and non-market ecosystem services of a novel combined food and energy agroforestry system in Taastrup, Denmark. Belts of fast-growing trees (hazel, willow and alder) for bioenergy production are planted at right angles to fields of cereal and pasture crops, and the system is managed organically with no inputs of pesticides or inorganic N. Field-based estimates of ecosystem services, including pest control, nitrogen regulation, soil formation, food and forage production, biomass production, soil carbon accumulation, hydrological flow into ground water reserves, landscape aesthetics and pollination by wild pollinators produced a total value of US$1074 ha^{-1} of which 46% is from market ecosystem services (production of food, forage and biomass crops) and the rest from non-market ecosystem services. Porter et al.[114] then extrapolated these values to the European scale and calculated that the value of non-market ecosystem services from this novel system exceeds current European farm subsidy payments. Obviously there are many challenges involved with using an ecosystem services or public goods approach to developing a support scheme for sustainable agricultural practices, but there has been much progress in the field of ecological economics recently and increased awareness at policy level of the potential of this approach[115].

Carbon credits. One particular area of environmental services where there has been more progress is the potential of an agroforestry approach to conserve and sequester C while maintaining land for food production and reducing deforestation and degradation of remaining natural forests. The 1997 Kyoto Protocol provides a mechanism by which countries that emit carbon in excess of agreed limits can purchase carbon credits from countries that manage carbon sinks. Leading the way with establishing tradable securities of carbon sinks to offset emissions, Costa Rica invested US$14 million in 1997 for the Payment for Environmental Services (PES), with 80% of funding coming from a tax on fossil fuels and 20% from international trading of carbon from public protected areas. This scheme led to the reforestation of 6500 ha, the sustainable management of 10,000 ha of natural forests and the preservation of 79,000 ha of private natural forests[49]. In 2003, the scheme was expanded to include agroforestry systems, and the Costa Rican government budgeted US$400,000 for the integration of agroforestry management into the C trading schemes with payments depending on the number of trees present on the farm[116]. Introducing carbon payments to landowners and managers of agroforestry systems in temperate regions

opens the way to obtaining additional income from these systems and may increase the attractiveness of establishing an agroforestry system, as well as adding value to established systems such as riparian buffers, shelterbelts, silvopastoral and silvoarable systems.

Conclusions

This review demonstrates that temperate agroforestry can provide an alternative approach to land management that, through the re-integration of trees and agriculture, can balance productivity with protection of the environment. With careful design and management, agroforestry systems can be more productive, and by producing a wide range of products, including food, fuel, fodder and timber, avoid the food or fuel conflict for land. Establishing trees on agricultural land can help to mitigate many of the negative impacts of agriculture, for example by regulating soil, water and air quality, supporting biodiversity, reducing inputs by natural regulation of pests and more efficient nutrient cycling, and by modifying local and global climates. The challenge now lies in promoting the adoption of agroforestry as a mainstream land use through research, dissemination of information and policy changes. In a second paper on temperate agroforestry, we discuss further how to develop modern systems that integrate ecological farming and agroecological advances to achieve sustainable intensification. We also discuss the existing barriers to wider adoption of agroforestry, and identify how these barriers can be overcome to promote agroforestry as a mainstream land-use system[117].

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Sharing a vision for biodiversity conservation and agriculture

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Abstract

Conservation biology and agriculture share a common landscape and a future that demands novel research and practice. Inevitably, limited resources create conflict in the absence of a shared vision forward. Therefore, given the similarities in proximate and even ultimate goals, we must envision a joint path toward renewable and resilient agroecosystems. In this commentary, I highlight the root of past conflicts and share a vision of progress forward that encompasses mutually beneficial outcomes. I include six areas of anticipatory research and inquiry at the intersection of conservation biology and agriculture to better identify shared goals and facilitate more frequent communication among disciplines.

Key words: agroecosystems, farmland, ecology, future collaboration, interdisciplinary

Climate change, population growth, biodiversity loss and broken biogeochemical cycles[1] are global challenges that necessitate a new and integrated vision for applied sciences, including agriculture and biodiversity conservation. Sharing a common landscape and resource base and inevitably a shared future, conservation biology and agriculture need to envision a joint path toward renewable and resilient agroecosystems where management, research and dialogue identify and work toward win–win situations. However, unlike other managed land-use systems such as forestry and range management, row crop agriculture has yet to embrace the conservation of biological diversity[2]. Likewise, perhaps due to the perception that row-crop farmland has little value for conservation efforts[3], conservation biology specialists have limited engagement with agronomists and farmers, choosing instead to focus on threatened and endangered species, protected landscapes and natural habitats. Yet for each to succeed, the definition of progress needs to take into account how conservation of biodiversity may increase productivity and profitability when farmers manage their systems in ways that augment regional biodiversity conservation priorities.

The poetic Wendell Berry[4] expresses a most critical barrier to this endeavor; 'My sorrow in having been for so long on two losing sides has been compounded by knowing that those two sides have been in conflict.' This inability to reconcile the conflict of competing land and resource use between biodiversity conservation and agriculture continues to impede significant progress toward addressing the above challenges. Ultimately, the failure to achieve a shared vision and one common language compounds the challenges and opportunity costs for researchers and practitioners alike.

At the root of this conflict is each discipline's dominant vision of how to singularly address each challenge. For example, global agricultural research has dramatically improved crop yields, in some cases tripling production. This undeniably benefited the near-term needs of producers and consumers[5]. However, these results are largely an outcome of biological simplification and control of regulatory ecosystem processes[6]. The subsequent unintended consequences for people and the environment, both locally and globally, are now being recognized[7]. In turn, conservation biology has begun to address the extinction crisis with respect to both deterministic drivers and the consequences of small populations[8]. Important local and federal conservation programs, including the Endangered Species Act, have successes to celebrate. Yet the field has focused its resources largely on protection and exclusion[9] or replicating historical habitat patterns as an end, largely ignoring the massive changes underway or that have already occurred in many ecosystems shaped by food production[10].

Although ecological and cultural barriers to multiple uses of arable cropland do exist, building a shared vision is necessary for progress to occur. It is perhaps the dissenting vision of conservation and agricultural specialists that provide the foundation. For instance, the organic food movement celebrates biodiversity as part of the farm

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system, whereas a new generation of conservation practitioners managing landscapes to increase the flow of ecosystem services gives credit to the value of those services provided by biodiversity in farmland[11]. Further progress and mediation of the perceived conflict will require more frequent interaction and discussion between disciplines focusing on how both fields can evolve beyond production and exclusion toward agroecosystem management that measures and balances multiple ecological functions.

As agronomy and conservation biology are driven and constrained by similar ecological and social forces[4], collaborative discussions with the goal of moving both disciplines forward simultaneously would prove fruitful. Building on the recent efforts of Sutherland, Pretty and colleagues to compile list of the 100 most important questions for each discipline[12,13], I describe six areas of anticipatory research and inquiry focused on the intersection between conservation biology and agroecology, where progress requires multidisciplinary teams of agronomists and conservation biologists. Examples of possible literature at the intersection of conservation biology and agroecology are included to stimulate this discussion.

1 The emerging discussion around managed lands maintained by human inputs of energy and materials and novel ecosystems embedded within managed ecosystems that have a diversity of species not occurring previously[14], provides an area of mutual interest to improve the dialogue between conservation and production. These 'neutral' areas provide a space beyond the protected areas and crop fields that are typically the focus of research and practice. Yet, the new mix of species[15,16] currently in these ecosystems may or may not be ecologically sound or beneficial. What are potential and realized costs and benefits of new biological communities and the resulting interactions and flow of ecosystem services among native, novel, managed and agroecosystems?
2 The acres planted with genetically modified crops are expanding. How will these unique new organisms interact with the existing species in an ecosystem? For instance, what are the implications of the transfer of novel traits to wild relatives or between non-organic and organic varieties? What are the consequences of limiting the available agro-biodiversity due to introduction of narrow genetically bred high-yield varieties and disappearance of many land races of crops and animals? Lastly, what measures do we use to assess the ecological impacts, both positive and negative[17,18]?
3 Sufficient, high-quality data are necessary to make informed decisions. How and what do we monitor in an ecologically meaningful way to understand the mechanisms, trade-offs and synergies encountered in agro and novel ecosystems? For example, conservation biology practitioners focus on biological measures of progress (e.g., species richness and population abundance), while agronomists focus on measures of agricultural resources (e.g., soil quality and crop yield). How should the researchers and practitioners prioritize and integrate the metrics of sustainability?
4 Production agriculture is essential to sustain a global population. Are there productive arable crop systems that mimic the ecological conditions necessary for biodiversity maintenance? What is the value of alternative agricultural land uses (e.g., biofuels[19] or perennials[20]) for biodiversity conservation?
5 Ecosystem services derived from biodiversity provide a variety of functional and economic benefits to croplands and society at multiple spatial and temporal scales[11,21]. In one case study, it is estimated that the economic value of pollination services provided by California wildlands is between $937 million to $2.4 billion[22]. How does the flow of ecosystem services differ between natural and agroecosystems? What are the economic benefits of ecosystem services at the farm scale as well as at larger spatial scales? The patterns of change in the flow of ecosystem services do not act in isolation[23]. What are the positive synergies between management for different ecosystem services? How can these services be bundled to aid management decisions and garner policy support?
6 Funds for agri-environmental schemes are often tied to multiple goals. For instance, the USDA Conservation Reserve Program was started as a means to address crop prices, but is now expected to retain soil and protect grassland birds, while farmers compete in the market. Because of the broad objectives, policy programs are often less successful than intended[24]. In addition, many mechanisms for farmland environmental programs are often perceived as rigid[25], or lack specificity regarding the objectives. The USDA National Organic Program standards, for example, define organic production as a system with practices that 'conserve biodiversity' yet the standards lack any rules or relevant measures. Thus, an important discussion is how the support dollars for nature conservation and farming should be integrated and allocated in the future.

We are currently at crossroads and yet there are many challenges in defining a mutual path forward[12,13,26–28]. Although the ultimate goals of food production and reduced extinction rates must be maintained, the proximate objectives of researchers, practitioners and policymakers can be redefined through collaborative discussions and evaluating the funding environment. One example of this progress is the work of the University of Nebraska-Lincoln (UNL) Organic Working Group with the recent grant titled 'Improving Organic Farming Systems and Assessing their Environmental Impacts'. As a multidisciplinary team, we have successfully integrated departments, defined common objectives, and worked with farmers to address local production and conservation goals. However, room remains for progress as we seek to define the appropriate scales and outcomes for research

and practice. Future discussions among practitioners, researchers and policy-makers should focus on mutual benefits and costs as well as processes and functions supported by biodiversity within and associated with renewable agroecosystems. Agriculture and biodiversity conservation need not be in conflict. Meeting shared goals through more frequent communication among disciplines can be the backbone for the success of future farming systems and nature conservation alike.

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Book Review

Cultivating a Movement: Excerpts from an Oral History of Organic Farming and Sustainable Agriculture on California's Central Coast. Edited by I. Reti and S. Rabkin 2011. UC Santa Cruz Library, Santa Cruz, California. 299 p., US $19.95, ISBN 9-780972-33431, paper.

Often the most compelling evidence for success of organic farming comes from the personal stories of farmers. Coupled with reports on the application of science in organics, the practical knowledge of people in the field provides a rich foundation for the ongoing growth of this intriguing sector of the food system. This collection of interviews by the staff of the Regional History Project is one unique activity of the UC Santa Cruz library, and a valuable contribution to the literature on organic systems.

In stark contrast to the industrial, large-scale monocultures that dominate the agricultural scene in California, 'tucked away along rivers, bluffs and canyons, and even within city limits, another, alternative agricultural landscape is emerging. The land tells this story through the voices of those who farm the soil and devote their lives to the sustainable agriculture and organic farming movement' (from the introduction by Congressman Sam Farr, D-CA). An unlikely complement to the high-tech production and global marketing systems, the organic food business continues to be the fastest-growing component of the food industry.

With focus on local production and direct sales to consumers, organic farmers often put high value in 'relationship marketing'. As the stories from these interviews reveal, the crops, systems and sales strategies are as varied and creative as the farmers who designed them. This diversity is reflected in the crops grown, the crop/animal systems and interactions, and the long-term plans of owners.

Crop diversity is most prevalent in the small vegetable and fruit farms designed to meet consumers' demand in local farmers' markets, in community-supported agriculture schemes, in farm shop sales and in marketing to local restaurants. The farmers practice crop rotations, use cover crops and integrate ruminants and other livestock to efficiently use crop residues and non-marketable products. Diverse spatial and temporal production strategies favor non-chemical integrated pest management and provide adequate nutrients to maintain soil fertility without importing any products from off the farm. The design of these self-sufficient systems leads to greater profitability than systems that depend on purchased organic inputs, many of which are expensive and have limited effectiveness.

Although most organic farms exhibit high levels of diversity, and certification requires rotations that include diverse species, some farmers opt for specialization in a few signature crops and animal products. Examples include mixed salad greens, heirloom varieties of specialties such as tomatoes, organic milk and cheese. The efficiencies of scale achieved on some farms allow sale to larger outlets such as Whole Foods, but most farmers opt to remain at a family scale or employ several workers or interns. Social responsibility looms large in most of the interviews in the book.

Another striking element of diversity emerges from the descriptions of farmers as well as researchers. Some have come from the mainstream, where bad experiences with farm chemicals encouraged them to seek other options. More often people have come from small, diversified farms or from the city, seeking a productive career and fulfilling lifestyle. Although sometimes long on ideals but short on prior experience, many new farmers arrive with energy, enthusiasm and innovative ideas about creating a profitable yet equitable food system for the future. Others were farm workers steeped in traditional crops and methods, or came from the traditional agricultural sector seeking independence as entrepreneurs in the food system. The interviews reveal a wide diversity of backgrounds, skills and motivations. It would be a serious error to generalize about organic farms as small, low-tech or traditional, although most are labor-intensive and clearly focused on local markets.

Agricultural organizations have emerged to support this movement in California. The United Farm Workers took on the complex issues of labor rights in agriculture. The Agriculture and Land-Based Training Association and the Rural Development Center in the Salinas Valley developed farming and business incubators for Hispanic farmers. Well known in academic circles is the UCSC Farm and Garden which has hosted apprentices on organic gardening and marketing since the early 1970s. Now the Center for Agroecology and Sustainable Food Systems, the center continues to enroll apprentices each year and maintains a modest research program with the Department of Environmental Studies. The Center conducts both student-oriented courses and extension

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programs through on-farm research, demonstrations, newsletter and field tours.

Cultivating a Movement is an interesting, thoughtful and well-written collection of reports on activities of real farmers, successful entrepreneurs and current academics. It is an inspiring book that provides a history of organic farming on the California coast, as well as a model for community building in organics that could be accomplished elsewhere.

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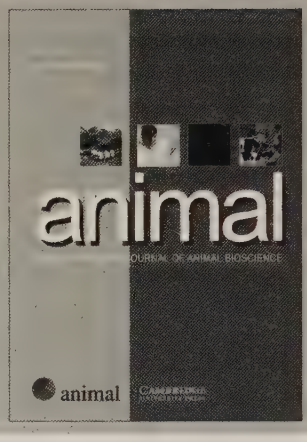

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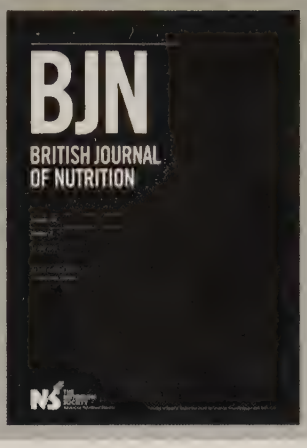

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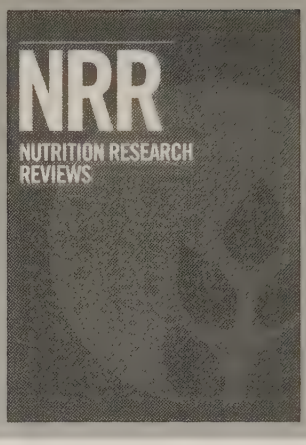

Published on behalf of The Nutrition Society

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Nutrition Research Reviews offers a comprehensive overview of nutritional science today. By distilling the latest research and linking it to established practice, the journal consistently delivers the widest range of in-depth articles in the field of nutritional science and has done so for the past twenty-one years. It presents up-to-date, concise and critical reviews of key topics in nutrition science, advancing new concepts and hypotheses that encourage the exchange of fundamental ideas on nutritional well-being in both humans and animals.

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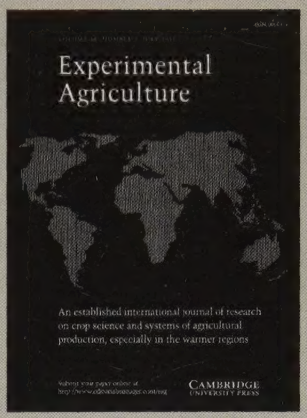
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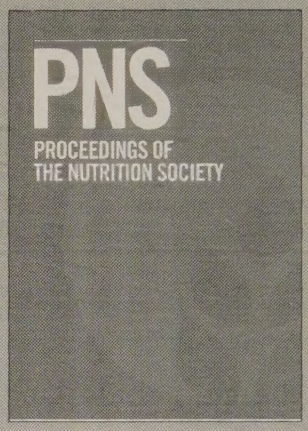

Proceedings of the Nutrition Society publishes papers and abstracts presented by members and invited speakers at the scientific meetings of The Nutrition Society. The journal provides an invaluable record of the scientific research currently being undertaken, contributing to 'the scientific study of nutrition and its application to the maintenance of human and animal health.'

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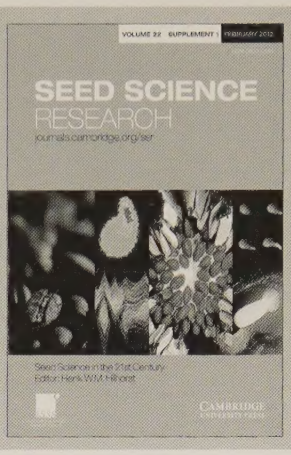

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